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16. ABSTRACT

Lake Sammamish, Washington, was studied during 1970-75 to determine its response to wastewater diversion in 1968. The results were compared with a pre-diversion study in 1964-65. Diversion reduced the phosphorus loading by about one-third (from 1.02 to 0.67 g P/m^2. yr and about 119 to 68 μ g/l in the inflow). Winter total phosphorus remained constant and no trend was shown in chlorophyll a in spring-summer. Water transparency remained the same.

Paleolimnological evidence suggests that the lake has been near its present mesotrophic state for about 100 years. This stability is thought to be due to the constancy of the water phosphorus concentration which is in turn controlled by the anaerobic-aerobic release and sedimentation of iron and its complexes.

Although the effect of a one-third reduction in phosphorus loading, as well as a similar increase resulting from the urbanization impact of the 1960's, was not apparent in the lake, without diversion loading would now be increased to 1.36 g P/m^2. yr.

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NUTRIENT DIVERSION: RESULTING LAKE TROPHIC STATE AND PHOSPHORUS DYNAMICS

by

Eugene B. Welch
University of Washington
Seattle, Washington 98195

Research Grant R 800512

Project Officer

Kenneth W. Malueg
Corvallis Environmental Research Laboratory
Corvallis, Oregon 97330

CORVALLIS ENVIRONMENTAL RESEARCH LABORATORY
OFFICE OF RESEARCH AND DEVELOPMENT
U.S. ENVIRONMENTAL PROTECTION AGENCY
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FOREWORD

Effective regulatory and enforcement actions by the Environmental Protection Agency would be virtually impossible without sound scientific data on pollutants and their impact on environmental stability and human health. Responsibility for building this data base has been assigned to EPA's Office of Research and Development and its 15 major field installations, one of which is the Corvallis Environmental Research Laboratory (CERL).

The primary mission of the Corvallis Laboratory is research on the effects of environmental pollutants on terrestrial, freshwater, and marine ecosystems; the behavior, effects and control of pollutants in lake systems; and the development of predictive models on the movement of pollutants in the biosphere.

This report describes the effects of wastewater diversion to a mesotrophic lake in Washington.

A.F. Bartsch
Director, CERL

ABSTRACT

Lake Sammamish, Washington, was studied during 1970-75 to determine the response to wastewater diversion in 1968. The results were compared with a pre-diversion study by Seattle METRO in 1964-65. Diversion reduced the phosphorus loading by about one third - from 1.02 to 0.67 g $P/m^2.yr$ and about 119 to 68 µg/l in the inflow. Winter total P remained constant at about 30 ± 2 µg/l and no trend was shown in chlorophyll *a* in spring-summer with a year-to-year average of about 7 ± 2 µg/l. Water transparency remained the same - a summer mean of 3.3 in Secchi disk depth.

Paleolimnological evidence suggests that the lake has probably been near its present mesotrophic state for possibly 100 years with some alteration during the late 19th and early 20th century logging activity in its watershed. A thorough survey in 1913 showed trophic state indices at similar levels as today. The lake has a history of relative stability in trophic state even though P loading was decreased by about one third and also increased by at least that much.

The lake's biological state did not respond to the P loading change largely because the water column P content did not change. Its stability is in turn thought to be controlled by the anaerobic-aerobic release and sedimentation of iron and its complexes. Plastic column experiments *in situ* showed rates of anaerobic release to range from 3-5 mg $P/m^2 \cdot day$ and measured sedimentation rates to exceed the release rates, with the greatest sedimentation occurring after turnover and the reestablishment of an aerobic water column, although as much as three-fourths of the water column P content could have come from sediments by the end of summer stagnation it was largely unavailable for spring phytoplankton because of subsequent sedimentation with iron.

The increase or excess in P loading during the greatest urbanization impact in the 1960's may well have gone to the sediment as was suggested by mass balance considerations if P sedimentation varies with P loading. Also the introduction of waste P into the major inflow stream 2 km above the mouth, rather than directly into the lake, argues for its sedimentation upon entering the lake. Without diversion, however, loading would now be at 1.36 g $P/m^2 \cdot day$ and probably some increase in water column P content in winter would be apparent.

Seasonal dynamic changes in P and chl *a* during spring were studied with a mathematical model that simulated those and two additional state variables. The model provided an interesting procedure for studying the mechanisms controlling the timing and maximum for the spring diatom bloom. The model should be extended to study the dynamics of P and Fe in the fall-winter period.

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SECTION 1
CONCLUSIONS

1. About 7,000 kg/yr of phosphorus was diverted from Lake Sammamish with waste water in 1968. This amounted to an aerial loading reduction of about one-third - from 1.02 to 0.67 g $P/m^2 \cdot yr$ or a reduction in mean flow weighted inflow concentration from 119 to 68 µg/l.
2. Continuous monitoring of the major inflow stream showed that daily observations were necessary to avoid missing 25-30% of the annual input of P. High concentrations of P (200-300 µg/l) and high flows occurred together on four separate days in 1973-74 and four in 1974-75 which amounted to such a high fraction.
3. The reduction in loading had no measurable impact on lake water concentration of total P and chlorophyll a, water transparency or hypolimnetic oxygen deficit rate. The blue green fraction of the phytoplankton did seem to have declined significantly - by over 40%.
4. Although most trophic state indicators did not change, the lake nevertheless remains mesotrophic with a mean summer transparency (Secchi disk depth) of 3.3 m and a maximum of near 6 m. This has been the state of the lake since early in this century and possibly since the settlement in the watershed of European man although logging had an impact in the late 19th and early 20th centuries.
5. Maintenance of lake trophic stability is most probably due in large part to the constancy of the winter P concentration, which in turn is controlled by rather high anaerobic release rates from the sediment (3-4 mg $P/m^2 \cdot day$) in summer, but more importantly high sedimentation rates following autumn turnover of usually about 4 mg $P.m^2 \cdot day$, but initially up to 14 mg $P/m^2 \cdot day$. Much of the P fluctuation was correlated with iron, which is no doubt controlling the P cycle in the lake - anaerobic release and aerobic sedimentation. Much of the higher pre-diversion P load evidently did not occur in the water column, was thus unavailable to algae, and only served to increase the sedimentation of P through the above and other processes.

6. Although the effect of a one-third reduction, as well as a similar increase, in P load to Lake Sammamish was not apparent, the diversion surely had a preventative value. Without diversion loading would now be 1.36 g $P/m^2 \cdot yr$ and could well have exceeded the stability mechanism(s) for water-column P.
7. A mathematical model with four state variables simulated P and chl a concentration reasonably well, but the model served primarily to study seasonal dynamics of the system. Although the work had only begun it has much promise for better understanding of the lake's cycling processes.

SECTION 2
RECOMMENDATIONS

1. Work should continue on the seasonal dynamics phase and associated modeling activity. With more continuously monitored key variables the carefully measured rates of P interchange with the sediments could be effectively utilized in the modeling process to determine the causative processes involved in maintaining such a steady lake water concentration of P in winter.
2. Effort should also include the nearshore area where changes in attached algal biomass should first appear as a result of the increasing urban development and its associated greater storm water input to the lake.

SECTION 3
INTRODUCTION

Since the turn of the century, cultural eutrophication of the Nation's waters has become a problem of ever-increasing magnitude. The rapid, sprawling growth of our communities has perturbed surrounding watersheds, augmenting the point and non-point sources of nutrient supply to many lakes. These artificial sources have stimulated primary production, often culminating in massive, foul-smelling plankton blooms that cloud the water. Typically, such events arouse adverse public reaction, with a resultant out-cry for corrective action.

The strategies and schemes for remedial action run the gamut from nutrient diversion to sealing the lake bottom with plastic sheeting. Perhaps one of the most frequently applied solutions has been to divert incoming nutrients away from an affected lake. The results can be very spectacular, as in the Lake Washington case (Edmondson, 1970, 1972), but also extremely expensive. The cost of the Lake Washington sewage diversion project exceeded $125 million (Gibbs, et al., 1972).

Subsequent to the Lake Washington project, the Municipality of Metropolitan Seattle (METRO) initiated a similar program for Lake Sammamish, which seemed to show early signs of movement to a eutrophic state (Isaac, et al., 1966). The secondary effluent from the city of Issaquah, Washington, and waste from a milk processing plant were diverted in September of 1968 at a cost of $3 million. Located only 3 kilometers east of Lake Washington, Lake Sammamish was expected to display a rapid improvement like its sister lake. To determine if the lake would respond to nutrient diversion and the extent and rate of such a response, monitoring of the lake has been carried out from 1969 to 1975. The Pre-diversion data on the lake consists of a two-year study by METRO (Isaac, et al., 1966) and a one-day survey by Kemmerer, et al. (1923) in 1913.

In addition to continuous monitoring of limnological characteristics, special studies of secondary production (zooplankton and fish), nutrient exchange rates between sediment and water, phytoplankton uptake of nutrients, feeding rates of zooplankton, profundal bottom fauna, and dynamic modelling of the phosphorus and nitrogen cycles have beer conducted, as well as a careful evaluation of the nutrient (particularly P) income (Birch, 1974; Emery, 1972; Hendrey, 1973; Horton, 1972; Monahan, 1974; Moon, 1973; Pederson, 1974; Sturtevant, 1974; Rock, 1974; Welch, et al., 1973; and Welch, et al., 1975).

Most of this effort has been for the purpose of estimating the change in the nutrient income, and defining observed changes in trophic state indicators and the processes that have permitted the lake to remain mesotrophic in spite of alteration of the P loading through the diversion project. In fact, core analyses have shown that several indicators of trophy have remained unchanged for over 100 years.

The diversion has subsequently been shown to have amounted to one-third of the lake's P loading. Apparently the lake's internal sediment-water interchange mechanism controlled by iron has resisted P loading changes over a range of at least 0.7 - 1.1 g $P/m^2 \cdot yr$. and allows the available water column P content to remain remarkably stable. This may well be the principal cause for the lake's lack of response to diversion. However, stability may result for other reasons and could not be expected to persist over a much greater range in loading and when viewed over the range of trophic states and loading that exist in the world's lakes, the range examined in Sammamish appears rather small.

For this project, the specific goals were to determine changes in water column concentrations of nutrients, phytoplankton chlorophyll *a* and species composition, and transparency following diversion of nutrients and determine, through measurement of external loading, the resultant fractional decrease in nutrient income. Further, the sediments were to be analyzed for N, P and C content and exchange rates between sediment and overlaying water determined in order to evaluate the relative

importance of the sediments as an internal source of P. Lastly, the understanding of the lake's behavior in response to changing nutrient input was to be extended by incorporating knowledge of the principal process rates into a mathematical model. Models of the simple input-output, steady state type described by Vollenweider (1969) and modified by Lorenzen (1973), as well as the multiparametric type (Tang, 1975), have been used. The steady state model application has verified the potential stability of the lake P content in response to changing P input while the multiparametric model has elucidated a more dynamic accounting of the P (and N) supply during the spring phytoplankton outburst.

A principal advantage in defining the response of Lake Sammamish to nutrient diversion that was cited earlier, is its proximity and comparability to that of Lake Washington. Although many characteristics such as geologic age, water quality type, meteorologic conditions and flushing rate are relatively similar, some are distinctly different. Lake Washington is essentially twice the depth as Sammamish, does not completely lose its hypolimnetic oxygen, while Sammamish does, and apparently is more subject to wind mixing. If not for those differing factors, the response of the two lakes to two fractions of diverted P could be considered with more validity. The diversions removed nearly three-fourths of the entering P from Washington, while only about one-third was removed from Sammamish. Whether the principal reason for no response in Sammamish is the control on water column P by the sediments or the smaller fraction of P diverted can be debated. The input-output model suggests that it is the former, while Vollenweider and Dillon (1974) indicate that >50% diversion of P should be necessary for improvement. To be sure, the year-to-year variation in the P input to Sammamish has been about equal to the fraction diverted. Probably a combination of the two is involved as will be emphasized in the report.

SECTION 4
GEOGRAPHIC DESCRIPTION OF LAKE SAMMAMISH

The waning of the Wisconsin glaciation (14,000 BP) left the Puget Sound lowlands dominated by striated hills, rolling uplands, and deeply cut troughs. Today one trough is occupied by Lake Sammamish, a second by Lake Washington with the meandering Sammamish River connecting the two. A mild, maritime climate now prevails, annually producing 90 centimeters of precipitation and a mean monthly temperature of 11.5°C (52.7°F). Direct sunshine is present 45 percent of the daylight hours. The lake is monomictic with stratification beginning in May and building to a maximum in August. The thermocline is completely eroded by late November and the lake remains homothermal until the following May. Table 1 provides a summary of the pertinent geographic conditions.

Table 1. SUMMARY OF BASIN GEOGRAPHY

Parameter	Lake Sammamish
Location	
Altitude (meters above mean sea level)	12
Longitude	122°05'W
Latitude	47°36'N
Size of Drainage Basin (km^2)	253
Duration of ice cover	none
Evapotranspiration (cm)	23.7
Evaporation (cm)	5.1
Precipitation (cm)	90
Maximum monthly precipitation (cm)	39
Minimum monthly precipitation (cm)	0

The predominant surface stratum of the drainage basin is a light, gray till. This till is a hard unsorted mixture about 46 meters thick, consisting of clay, sand, silt, and gravel. Although the till is relatively impermeable, thin beds of sand and gravel commonly yield small quantities of perched water. Aquifers transect the basin, with several artesian wells surfacing within the basin (Liesch, et al., 1963). Coal seams are located in the southern half of the watershed, while high quality sand and gravel, refractory grade clay, quarry basalt and cinnebar deposits are scattered throughout the basin (Livingston, 1971).

A geologic cross-section cutting through Issaquah in an east-west direction shows base rock consisting of marine sedimentary rocks on the west side of the Lake Sammamish valley. On the east side is volcanic rock with overlying layers of clay, advanced stratified drift, till and sedimentary deposits (Liesch, et al., 1963).

Prior to the arrival of European settlers in 1862, the Lake Sammamish basin was covered in a climax formation of Western Red Cedar (Thuja plicata), Western Hemlock (Tsuga heterophylla), and Douglas Fir (Pseudotsuga taxifolia) (Hansen, 1938). Heavy logging around the turn of the century left the basin in second growth forest. Today 80% of the watershed remains in second growth, primarily red alder (Alnus oregona) with scattered maple (Acer, sp.) and willow (Salix sp.). The significant historical events in the basin are listed in Table 2.

The population of the basin has grown from three families in 1862 to the present 40,000, the majority of the growth coming in the last 10 years. The only sizeable concentration is located in the town of Issaquah, population 4,500. The town is comprised of the small businesses required to support a residential community. The only industrial development is a dairy processing plant and a state salmon hatchery. Within the watershed are several gravel operations and a county sanitary landfill. Large residential developments have been built throughout the entire west side of the lake. The east side is dotted with small farms, but the major portion of the land remains in second-growth. A narrow

Table 2. HISTORICAL EVENTS OF INTEREST IN THE LAKE SAMMAMISH BASIN (AFTER FISH, 1967).

Date	Event
1862	Three families settled along lakeshore
1864	Three settlers and four Indians killed in uprising
1868	Hops started and become principal agricultural crop
1870	Census listed 28 men; 24 were farmers
1880-1920	Logging very important, lake often covered with logs
1887	Large scale coal mining (1913-14 boom years), dies out in 1920's
1888	Railroad built around eastern shore
1889	Donnelly mill closed (large sawdust pile on lakeshore), large mill located at Monohan (pop. 200)
1900	Hops no longer farmed
1912	Sammamish Slough dredged, shortened from 19 km to 10.5 km
1913	Issaquah Sewerage Agency formed
1925	Town of Monohan burned down
1940	Secondary treatment (trickling filter) built for 0.15 MGD capacity
1960	Population of Issaquah is 1,870
1968	Treated effluent diverted from Issaquah Creek
1970	Population of Issaquah is 4,314

strip of land along the east shore of the lake has been subdivided into residential tracts. The upper valley drained by Issaquah and Tibbets Creeks is primarily forested with scattered farms and small clusters of houses (Figure 1).

Lake Sammamish State Park, located at the south end of the lake, draws over 600,000 visitors each summer. Two commercial and an undeveloped county park are also located on the lakeshore. In addition, a large county park touches the northern end of the lake and extends along the Sammamish River. Both Pine and Phantom Lakes have county Parks located on their shores. Presently the waters of Lake Sammamish support a variety of water sports, including water skiing, pleasure boating, boat racing, swimming and fishing, besides providing an aesthetically pleasing background for picnics and related activities carried out at the several lakeside parks.

The primary point sources of wastewater within the basin were the town of Issaquah, the milk processing plant, and the fish hatchery. Since 1968, the effluent from the town's trickling filter plant (568 m^3/d) and the milk plant (284 m^3/d) have been diverted out of the basin. Today only the milk plant cooling water (227 m^3/d from groundwater) and the hatchery passthrough water, which originally comes from Issaquah Creek, are discharged to Issaquah Creek, and to Lake Sammamish. Only the sparsely settled east side and upper valley sections of the watershed remain on septic tanks. Hence the percentage of the basin population on septic tanks is small.

PALEOLIMNOLOGY

Paleolimnological evidence suggests that Lake Sammamish has maintained a trophic state within the mesotrophic range for at least the past 100 years in spite of settlement in the watershed by European man and an increase and decrease in the phosphorus income of at least one-third. The only shift in the 150 years of paleolimnologic history shown in the organic matter profile (Fig. 2) from Lake Sammamish is from 1880 to 1910, using a sedimentation rate of 3.5 mm/yr determined from a recent historical event. This

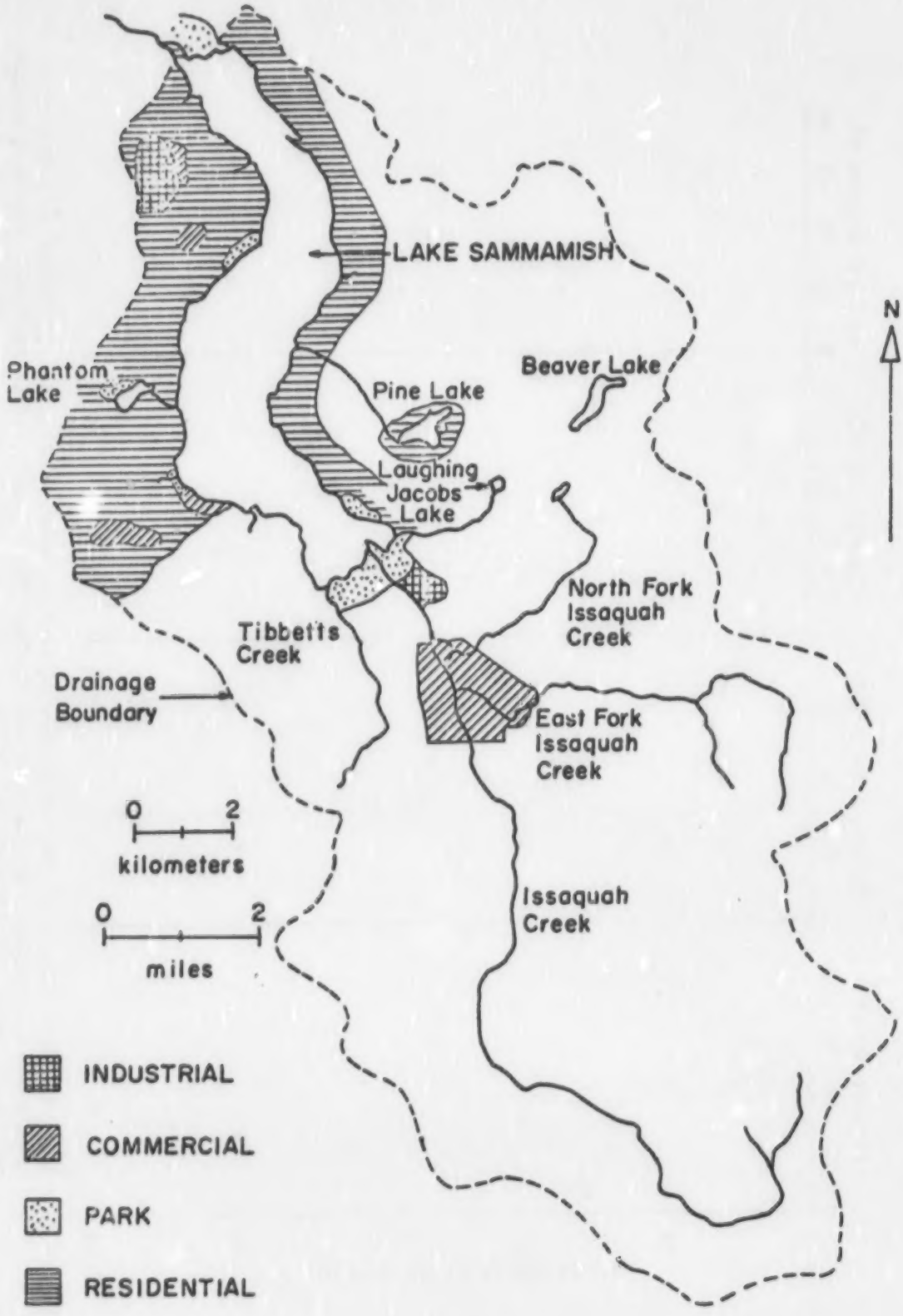


Figure 1. Land Use Map of the Lake Sammamish Watershed.

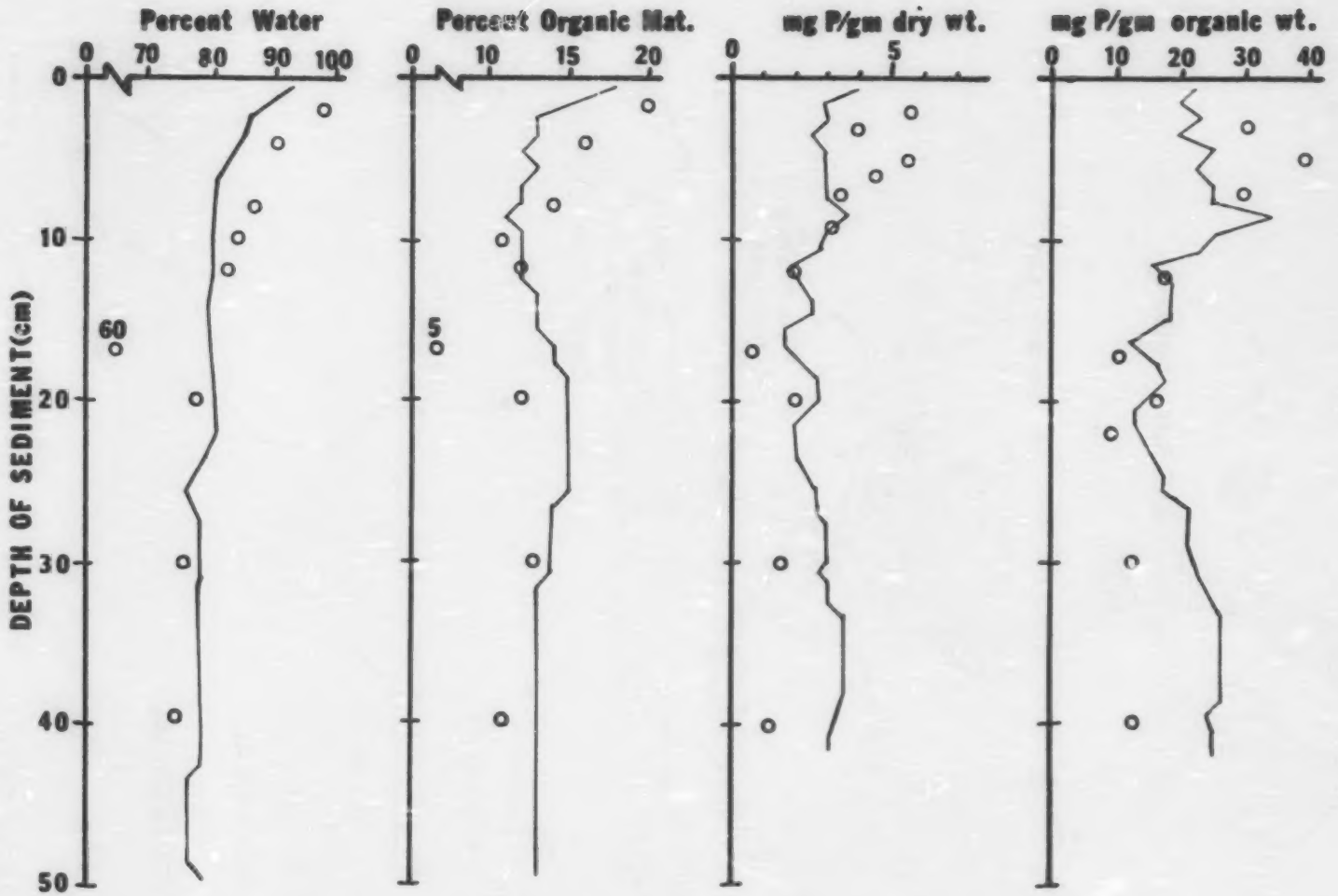


Figure 2. Vertical distribution of percent water, percent organic matter, and phosphorus as a fraction of the dry weight and organic matter of the sediment. Open circles represent values from Lake Washington cores (Shapiro et al., 1971).

bulge corresponds to the period of intense logging in the Pacific Northwest and a shift in pollen from confers to alders. The organic matter bulge probably resulted from sawdust and bark deposits from mill and log rafting activities. If a change from aerobic to anaerobic conditions had occurred during this period it surely would have been evident in the organic matter content.

Comparison with Lake Washington results (Shapiro, et al., 1971) shows that both phosphorus and organic matter content were higher than levels in Lake Washington in early years, lower than Lake Washington in recent years, but remained relatively constant over the last century (Fig. 2). The sediments in Sammamish clearly show less impact of man than in Lake Washington and are characterized most strongly by the constancy of their contents.

In a more detailed analysis of historical sedimentation using Pb^{210} dating, four distinctly different rates were observed with the highest rate having occurred between 1932 and 1944 following logging activity. Although the concentration of P in the sediment actually decreased, the loading rate to the lake during that period may have been four times the present loading if the increased rate of sedimentation is considered. The increased sedimentation rates are thought to be a result of increased erosion following deforestation, which was at a peak during 1910-1930 (Birch, 1976).

A one-day study of the lake by Kemmerer, et al. (1923) in August, 1913, strongly suggests that the lake was anaerobic and mesotrophic then. Phytoplankton species, Secchi disk depth, and except for the absence of Diaptomus, zooplankton species were all similar then to those found in the lake today.

The pennate/centrate ratio of diatoms in the lake's sediments have also remained relatively stable over the past 100 years (Table 3). According to Stockner (1971) these values are within the 1.0 - 2.0 range for mesotrophy. For more details of the paleolimnological results see Rock (1974). Also, Wiederholm (1976) has found chironomid species that indicate mesotrophy throughout a 50 cm deep core and a 100 year history.

Table 3. DIATOM RATIOS FOR LAKE SAMMAMISH CORE 08.

Date	Pennate/Centrate
1972	1.2
1955	1.0
1940	1.5
1925	1.1
1910	1.2
1895	1.1
1880	1.5
1865	0.9
1850	0.9

Thus, it appears that the trophic state of Lake Sammamish has remained relatively unaffected by activities in the watershed since the settlement of European man. The associated increases in P income during this period have been at least on the order of 30 to 40 percent and no doubt much greater considering all potential point and non-point sources.

SECTION 5

MORPHOMETRIC AND HYDROLOGIC DESCRIPTION OF LAKE SAMMAMISH

Lake Sammamish occupied a 13 km section of the Sammamish River Valley after the Wisconsin glaciation when the retreating Vashon glacier left a terminal moraine blocking the valley. Today the lake level is controlled by a wier at the head of the Sammamish River. The deepest section of the lake is located less than 3 km from the south end. If 15 m is considered as the division between deep and shallow, the surface area ratio of deep to shallow water is

$$\frac{12\ km^2}{7.8\ km^2\ (shallow)} = 1.54$$

while the ratio of epilimnion to hypolimnion volume is 0.99.

The study of water currents has been limited to the movement of Issaquah Creek water in the lake (Moon, 1972). During the period of winter mixing the creek water dispersal is primarily influenced by wind direction and velocity. The water was sufficiently dispersed at a distance of 500 m to make the tracer undetectable. Similar studies made during thermal stratification showed the creek water plunging into the metalimnion and dispersing in a fan-like pattern. Additional morphometric and hydrologic data are summarized in Table 4.

Waste-water effluent from Issaquah entered Lake Sammamish through Issaquah Creek. In winter, during high flow waste water P entered along with high levels of suspended sediment. In summer the waste P followed the Issaquah Creek inflow water which probably entered the open water area at depth.

Table 4. SUMMARY OF PERTINENT HYDROLOGIC AND MORPHOMETRIC CHARACTERISTICS FOR LAKE SAMMAMISH.

Parameter	Lake Sammamish
Surface Area of Lake (km^2)	19.8
Drainage Area (km^2)	253
Lake Volume (km^3)*	0.35
Depth	
Mean (m)	17.7
Maximum (m)	32.0
Epilimnion (m)	8.8
Euphotic (m)	7.3
Width	
Mean (km)	1.5
Maximum (km)	2.4
Length of Lake (km)	13.0
Length of Shoreline (km)	34.0
Water Retention Time (yrs)	1.8
Stream Inflow (km^3/yr)	0.198
Stream Outflow (km^3/yr)	0.203
Groundwater Infiltration (km^3/yr)	0.0
Groundwater Exfiltration (km^3/yr)	0.01
Duration of Stratification (mos.)	7

*influenced by wier

SECTION 6
LIMNOLOGICAL CHARACTERIZATION

TEMPERATURE

Lake Sammamish is a monomictic lake that begins thermal stratification in May. Maximum water column stability occurs by late August and destruction of the thermocline is complete by late November. The temperature range is from a minimum 5.5°C to a maximum 25.5°C.

LIGHT

The depth of visibility has been determined by means of the Secchi disk. The annual mean for the six years of data is 3.3 m. The lowest seasonal mean (3.0m) occurs in the winter due to turbidity from the winter mixing and runoff. The springtime mean is only slightly higher (3.1 m), but these low values are due to the diatom pulse. The light penetration increases during the summer (3.5 m) and reaches its highest values in the autumn (3.6 m). The maximum Secchi disc measurements usually occur in September, the deepest recorded being 6.1 m.

Light extinction was determined by a submarine photometer. The bottom of the euphotic zone was considered to be at a depth receiving 1% of the surface light intensity. The mean depth of the euphotic zone is 7.0 m, while the range is from 5.0 to 12.5 m.

ALKALINITY AND pH

Lake Sammamish has a pH range of 6.3 to 9.6 due to biological activity. Correspondingly the alkalinity as $CaCO_3$ ranges from 26 mg/l (0.52 meq/l) to 42 mg/l (0.84 meq/l), while the mean is 33.3 mg/l (0.67 meq/l).

DISSOLVED OXYGEN

During the winter, the oxygen content essentially remains at an air saturation level, approximately 12 mg O_2/l, due to continual

circulation. The development of thermal stratification in early May results in a clinograde O_2 curve that approaches zero oxygen levels (0.1 mg O_2/l) in the bottom waters by late July or early August. The hypolimnetic oxygen deficit continues to increase until early October. By this time the entire hypolimnion (below 15 meters) has less than 1 mg O_2/l. Oxygen levels start to increase with the coming of the autumnal circulation.

The depletion of oxygen in the hypolimnion is dramatically shown in the calculation of hypolimnetic oxygen content in early October (Table 5). Also shown are "actual" deficit rates. The "actual" rates are based on oxygen saturation of 12 mg O_2/l at 7.5°C occurring on April 15th and the oxygen present at the end of stagnation.

Table 5. HYPOLIMNETIC (DEPTH GREATER THAN 15 m) OXYGEN CONTENT AND "ACTUAL" OXYGEN DEFICIT RATE.

Year	Total Hypolimnetic O_2 present at start of stratification (metric tons)	Total Hypolimnetic O_2 present at end of stratification (metric tons)	Number of days of stratification	O_2 Deficit Rate (mg O_2/ cm^2-day)
1970	1120	21.3	178	0.051
1971	1120	12.5	192	0.047
1972	1120	45.7	176	0.050
1973	1120	21.3	185	0.049

MAJOR AND MINOR IONS

The results of a single survey are shown in Table 6. Neither $SO_4^=$ nor Cl^- have been measured, while the only trace metals measured have been Mn^{++}, Zn^{++} and Pb^{++}

Table 6. Trace Metal Survey on October 24, 1971 for Lake Sammamish

Location	ppm				ppb			
	Ca	Mg	Na	K	Fe	Mn	Zn	Pb
Lk. Sam. 612 Surface	12.80	3.42	8.43	1.01	40	19	376	0.5
8 m			8.47	0.98	63	40	300	0.5
16 m	8.40	3.44	8.15	0.94	280	600	35	0.8
25 m	8.95	3.68	8.17	1.00	1020	1660	34	0.8
Inflows Issaquah Ck.	12.40	3.70	9.31	0.94	450	35	318	0.9
Tibbetts Ck.	24.70	8.15	14.59	1.52	110	20	150	4.2
Outflow Sammamish River	6.05	3.00	8.16	1.13	25	9	7	0.6

SECTION 7
METHODS AND MATERIALS

LAKE MONITORING

The lake water column has been monitored at a central location from 1970 through 1975 by students and faculty in the Civil Engineering Department, University of Washington. Prior to the 1968 wastewater diversion the Municipality of Metropolitan Seattle (METRO) monitored the water column at the same location for 1.5 years from July 1964 through 1965. Thus, the effect of diversion is evaluated primarily from the changes in trophic state indicators determined at that central station designated as 612 in Figure 3.

Sampling frequency has varied from monthly to weekly throughout the period, with frequency generally increasing in recent years. Twice monthly was generally considered necessary particularly during the growing season. After examining the 1970-71 data, Swayne (1973) showed that a frequency greater than 20 days between sample collections would result in an artificial smoothing in the temporal dynamics of many of the variables.

One station was considered representative. Emery (1972) showed that there was no statistically significant difference between chemical and biological variables at station 612 compared with a northerly located station indicated by METRO as 611 (Fig. 3).

During 1970-73 the photic zone was sampled at four depths based on the amount of light received. The depths corresponded to 95, 60, 30 and 1 percent of incident light. Secchi disk depths were used to locate depths for given light levels based on a nomograph determined by Emery (1972). This was largely for the purpose of insuring that maximum photosynthetic rates were measured regardless of incident light variation. The maximum in Lake Sammamish was found to vary from 30-60% of incident light. Samples for chemical analysis were also collected in

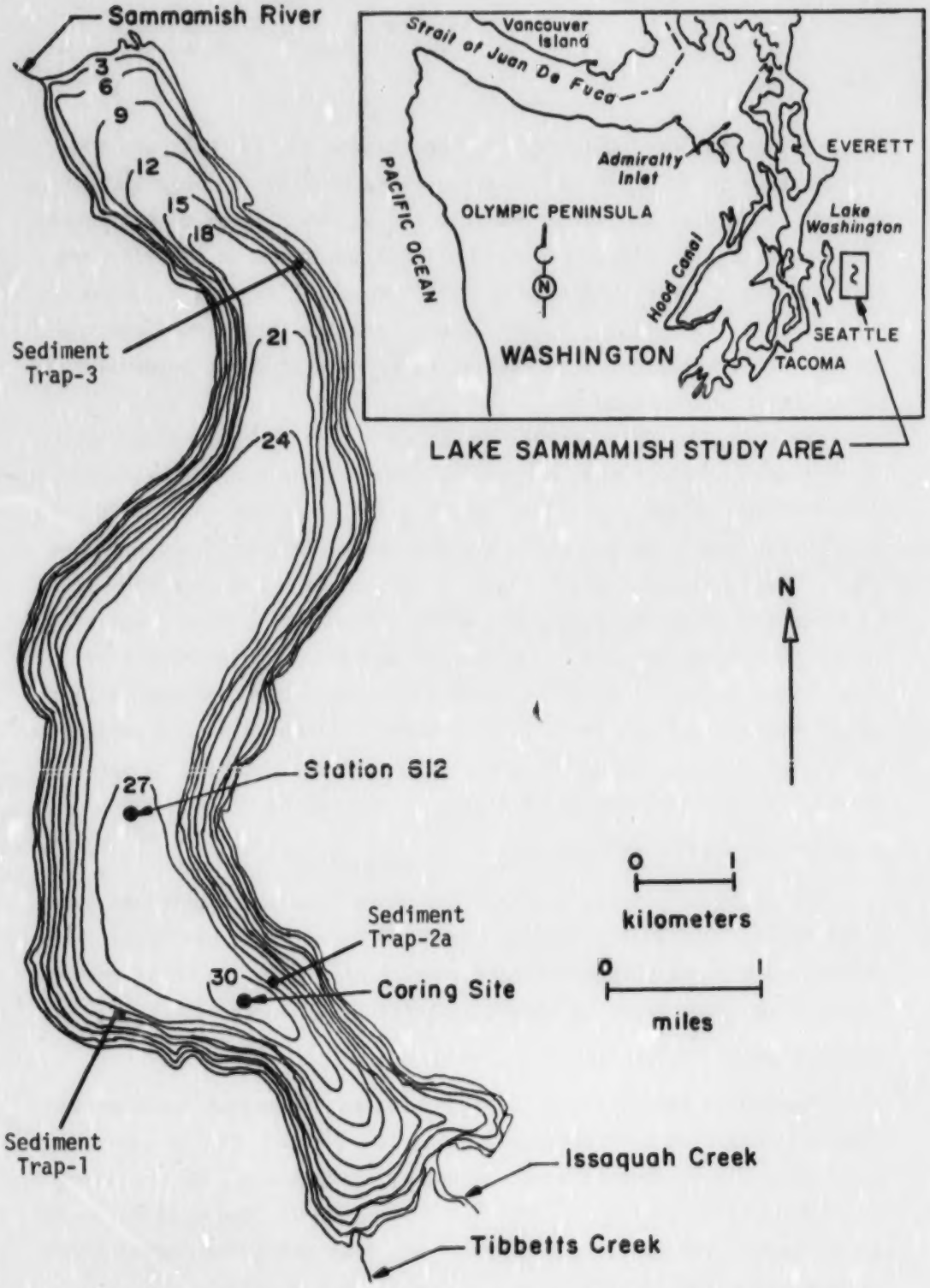


Figure 3. Location of sampling stations and bathymetric map for Lake Sammamish.

the hypolimnion, one half the distance between the 1% depth and the bottom and 1 meter above the bottom - a depth of 28 m. This station was not located at the lake's maximum depth, which is 31 m, but rather at 28 m (Fig. 3). Procedures were slightly modified in 1974-75 when specific depths were regularly sampled throughout the water column - surface, 5 m, 10 m, 15 m, 20 and 22 m. Also, in situ oxygen and temperature determinations were made at more frequent depth intervals than other chemical measurements.

The variables determined, the methods used and respective sources for the methods are shown in Table 7. The only method not explained is phytoplankton volume, which has varied with investigations; METRO used an inverted scope sedimentation along with specific cell measurements, Emery (1972) used an upright scope, centrifugation and specifically measured each observed alga cell, while subsequent work has been with an upright scope, but with a sedimentation method in 1973 and a millipore filter method in 1974-75. Both later sets were developed with the use of average volumes for a given species. Although direct comparisons of volumes are not comparable the percent volumes occupied by various groups, for example blue green algae, is considered valid.

IRON AND PHOSPHORUS AT OVERTURN

The relationship between iron and phosphorus was studied at overturns in 1972 and 1973. Samples for total P total Fe and oxygen were determined at weekly intervals and every 2 meters in the water column. Total Fe was determined by atomic absorption (APHA, 1971).

NUTRIENT INCOME ESTIMATION

Phosphorus and nitrogen loading into Lake Sammamish has been estimated by measuring the concentration of total P and total N (only NO_3 in water year 71) in several minor tributary streams. METRO monitored 9 in 1964-65 (Isaac, et al., 1966) and 12 in 1970-71 (Moon, 1972). The major surface inflow--Issaquah Creek--was also monitored during those periods and again in 1972.

Table 7. SUMMARY OF ROUTINELY MONITORED PHYSICAL, BIOLOGICAL AND CHEMICAL PARAMETERS

Parameter	Method	Reference
Physical		
Temperature Profile	YSI and Hydrolab Temperature Probe	
Light Penetration	Secchi Disk	
Euphotic Zone	Submarine Photometer	
Biological		
Primary Production	^{14}C Light-Dark Bottles	Strickland and Parsons (1968)
Phytoplankton Volume	Variable Sample	
Chl *a*	Extraction with 90% Acetone	Strickland and Parsons (1968)
Chemical		
Dissolved Oxygen Profile	Azide Modification-Winkler	American Public Health Assoc. (APHA) et al., (1971)
Total, Ortho-phosphate P	Molybdate Complexing Reaction	Strickland and Parsons (1968)
Nitrate, Nitrite N	Cadmium-Copper Column	Strickland and Parsons (1968)
Silicate	Molybdate Complexing Reaction	Strickland and Parsons (1968)
pH, Alkalinity	pH Meter	APHA et al., (1971)
Total N	UV Light Oxidation	Strickland and Parsons (1968)

sampled the streams once and Moon twice per month for nutrient content and flow, except that Issaquah Creek was continuously gauged by the U.S.G.S. at a point about 1.9 km from the lake. Moon collected 37 samples from Issaquah Creek in 1970-71 and 30 in 1971-72 (WY 71 and 72). Issaquah Creek was shown by Moon (1972) to carry about 70% of the inflow of P and 57% of the N (NO_3).

Because of the variability in nutrient budget estimates from person to person and by different means of data handling (see Rock, 1974), continuous monitoring of the Issaquah Creek nutrient content was begun in the 1973 water year. An automatic sample collector was installed in the U.S.G.S. gauging hut and samples were collected every eight hours for nutrient analysis. Over 1000 samples were provided for analysis from Issaquah Creek in the 73 WY. This insured that periods of short term peak runoff, in which total P was known to vary greatly, could be accurately determined. Daily collections were also made in the 1974 water year for total P and total N content.

In Moon's (1972) water budget for the lake, subsurface was assumed to contribute relatively little water to the lake in comparison to surface stream flow. Thus, the groundwater contribution of nutrient was also ignored in the budgets. Atmospheric input with rain to the lake surface was included, however, by monitoring the rainfall, albeit rather infrequently, during the 71 WY. The final significant source is the Washington Department of Fisheries salmon hatchery, which was briefly evaluated and included in the below mentioned special study.

To help resolve the problem of how much nutrient was actually diverted in 1968, a joint study was undertaken with METRO to reevaluate the contribution of the Darigold milk processing plant and the Issaquah sewage treatment plant effluents to the pre-diversion loading to the lake.

SEDIMENTATION

Sediment raining down from the epilimion was collected in traps located near the bottom at three additional stations besides the control 612 station. Sediment was collected by four 10 cm diameter plastic funnels which channeled the material into 50 ml centrifuge tubes. These four units were secured to a 30 x 30 cm polyvinyl chloride platform. Traps were suspended at various depths in the lake, with emphasis here on results from the bottom traps, 2 m above the bottom at all four stations.

The traps were located after collection intervals ranging from one week to several months by means of buoys submerged 1-2 m below the surface. This technique successfully avoided interference by the boating public. To minimize the decomposition of material, 5 ml of chloroform was added to each centrifuge tube as a preservative. A disadvantage of this technique is that congregating zooplankton are trapped and killed and then must be removed before analysis by filtration (0.5 mm) and then detritus particles > 0.5 mm must be removed from the net and added back to the sediment samples.

Sediment analysis consisted of centrifugation at 9000 rpm for seven minutes, decanted, dried at 60° C for 24-36 hours and weighed. This was begun immediately upon returning from the field. Prior to this, samples were examined microscopically to evaluate the qualitative nature of the material as to allochthonous or autochthonous origin.

Subsamples of dried sediment (20-100 mg) were analyzed for C using a Leco semi-automatic carbon analyzer and for N using the semi-microkjeldahl method (Bremner, 1960). Total P was determined according to the ascorbic acid-molybdenum blue method in Standard Methods (1971) on digested sediment. Sediment (20-100 mg) was digested in 10 ml Teflon crucibles by incubating for 12 hours at 120° C with 5 ml of 40% HF. Then 5 ml of concentrated HNO_3 was added and treated at 240° C for 1-2 hours. Oxidation was completed by further addition of 5 ml HNO_3 and cautious addition of 5 ml of 70% $HCLO_3$. Just prior to evaporation samples were cooled, removed with 1 ml concentrated HCL, filtered through #4 prerinsed Whatman paper and diluted with 50 ml demineralized, deonized water.

SEDIMENT-WATER INTERCHANGE

Phosphorus release from the sediment in Lake Sammamish was determined *in situ* by using six plastic columns inserted into the sediment in the vicinity of sediment trap 1 (Fig. 3). Two opaque columns and one transparent column were placed at depths from 10-12 m and two transparent and one opaque column were placed at 6-8 m depths.

Each column was 3.7 m long with a weighted base plate attached 0.6 m above the bottom. Volumes varied from 42 to 62 ℓ depending upon diameters. Sampling ports (1.3 cm dia) were located at 0.3, 1.9 and 2.8 m above the sediment. Tygon tubes 15 m long were extended from each port to a surface float. Each column was equipped with a plexiglass piston that allowed for the extraction of sample water and prevented the influx of oxygenated water and avoided the formation of a vacuum inside.

After samples were extracted deoxygenated water was forced back into the columns. About 5 ℓ or 10% of the column volume was removed with each set of samples.

Analytical procedures for P, Fe, DO, pH and alkalinity were the same as previously indicated.

Experiments were conducted at various time intervals over a three-year period. Usually, columns were sampled every 10-14 days. Effort was made to maintain the column samples under anaerobic conditions by collecting the water in chambers previously purged with N_2. However, when filters clogged and were changed O_2 was introduced thus biasing the soluble P results.

The columns similated two conditions in the lake. The opaque columns represented the dark hypolimnetic water while the transparent ones represented the processes in the euphotic zone.

The experimental periods in 1973-74 were May to July and July or August to December or January. For the early experiments the duration ranged from 63-76 days for three columns. For the later experiments the periods ranged from 121-171 days. In some instances the columns were purged with N_2 to immediately exhaust the DO while in others DO was allowed to exhaust naturally via respiration.

DYNAMIC MODEL

The model developed for Lake Sammamish is made up of two major sub-models, a mixing model, which computes the daily mixing depth and temperature profile, and a phsophorus model, which utilizes the physical, chemical and biological parameters pertinent to the lake. The model determines the magnitude of the pertinent state variables.

Mixing model

The mixing model computes the temperature profile by first calculating the thermal energy input to the surface waters considering insolation, conduction, evaporation and back radiation. The temperature throughout the water column is then determined by summing the temperature value from the previous day at each depth with the effect of the diffusivity of energy within the water column at that depth.

The mixing depth is then computed by determining the point in the water column where the kinetic energy provided by the wind shear on the lake's surface coupled with the convective mixing energy due to instability caused by temperature differential, is equal to the potential energy of the thermal gradient integrated from the surface.

The last operation the mixing model performs is to average the temperature from the surface to the mixing depth and adjust the temperature at each depth within this zone to an average value.

Phosphorus

To facilitate modeling, the lake was divided into three zones, the epilimnion, defined as the layer between the surface and the mixing depth, the hypolimnion, defined as the layer between the mixing depth and one meter above the bottom, and the sediment-water interface zone, defined as the layer containing the bottom sediments and the water one meter above the bottom. From these definitions it is observed that on days when complete mixing occurs the hypolimnion does not exist.

The phosphorus model has four state variables: phytoplankton (X1); ortho-phosphorus (X2); zooplankton (X3); and detritus (X4). These state

variables are represented by the pools (ovals) in Figure 4. The equations for the changes in those state variables over time are also shown in Figure 4.

Table 8 lists the subprocesses which make up the functional equations for the state variables. The notation used for system constants and variables is listed in Table 9 along with their definitions, dimensions and the values for the constants. As may be readily observed from Table 8, all of the subprocesses, with the exception of photosynthesis (Photo), are either Michaelis-Menton type or logistic functions.

The arrows in Figure 4 show the model processes in graphic form including sources, sinks and the exogenous variables (driving functions) which are mixing depth (H), temperature profile (T[I]), incident solar radiation (RO) and inflow (Q_{in}). Outflow is assumed equal to inflow in this model and is also notated Q_{in}. The sediment acts as a source and/or sink for the phosphorus depending upon physical conditions.

The phosphorus model was developed as a differential equation model using a fourth-order Runge-Kutta technique to approximate the time dependent changes in the state variables on a daily basis. The model is presently implemented in both its original form and as a difference equation model using a variable time step to insure that no state variable changes more than five per cent (5%) in any one time period.

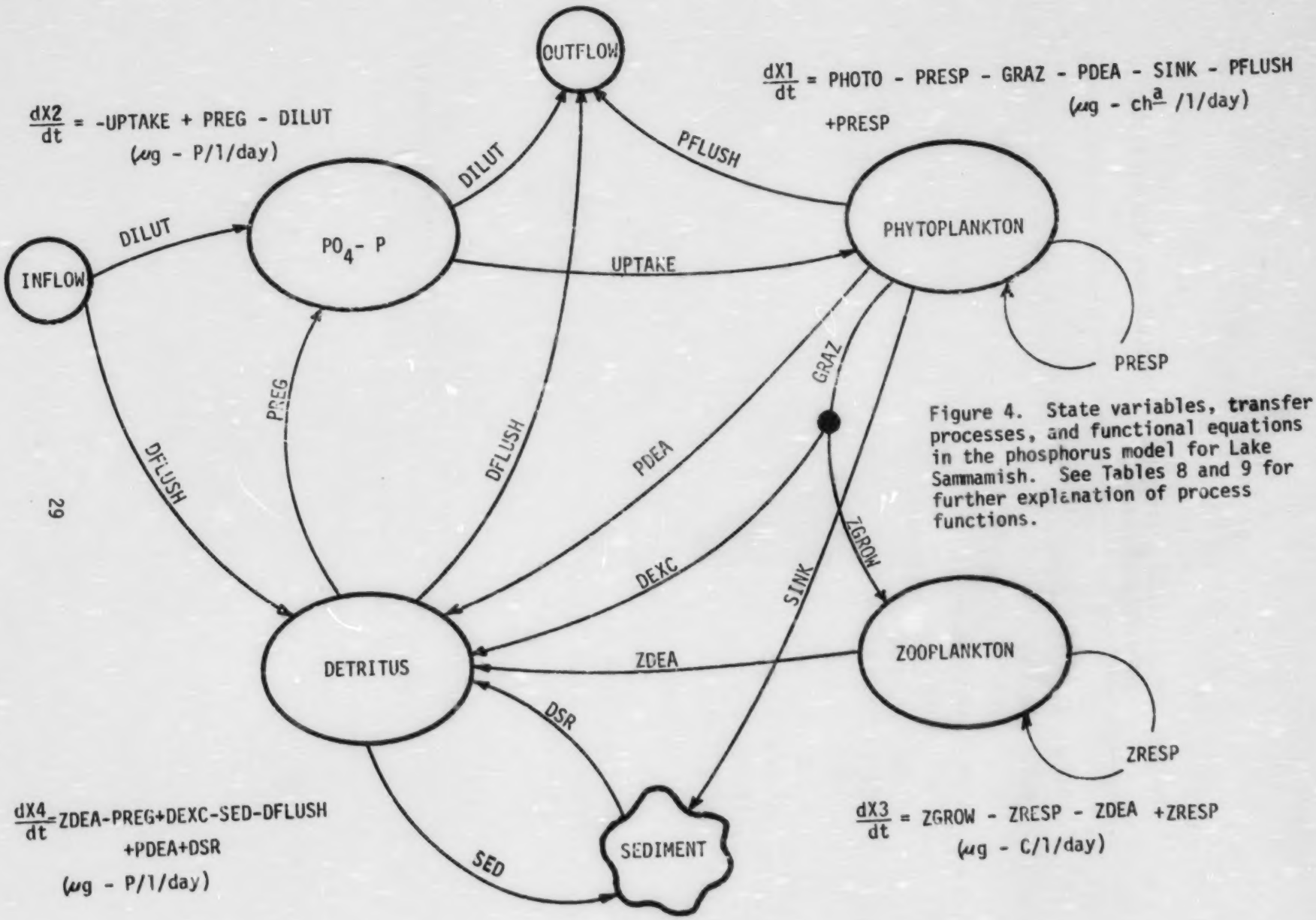


Figure 4. State variables, transfer processes, and functional equations in the phosphorus model for Lake Sammamish. See Tables 8 and 9 for further explanation of process functions.

Table 8. MATHEMATICAL EXPRESSIONS OF INVOLVED SUBPROCESSES IN THE PHOSPHORUS CYCLING MODEL.

$$PHOTO = k1 \cdot e^{(T/10)} \cdot (2R/RO \cdot dz) e^{(1-2R/RO \cdot dz)} \cdot (X2/(k2+X2))X1$$

$$PRESP = k3 \cdot T \cdot X1$$

$$GRAZ = k4 \cdot X3(X1/(k9+X1))$$

$$SINK = k5 \cdot dX1/dz$$

$$PDEA = k10 \cdot X1$$

$$PFLUSH = Q_{in} \cdot X1/V$$

$$UPTAKE = CO \cdot CO1(PHOTO)$$

$$PREG = k7 \cdot X4$$

$$DILUT = (.25 \cdot k6 - X4)Q_{in}/V$$

$$ZGROW = CO1 \cdot CO2 \cdot GRAZ$$

$$ZRESP = ZRES \cdot T \cdot X3$$

$$ZDEA = k8 \cdot X3$$

$$DZD = CO \cdot ZDEA$$

$$DEXC = (1-CO2) \cdot CO \cdot CO1 \cdot GRAZ$$

$$SED = k11 \cdot dX4/dz$$

$$DFLUSH = (.3 \cdot k6 - X4)Q_{in}/V$$

$$DPZ = CO \cdot CO1 \cdot PDEA$$

$$DSR = k12 \cdot A_{sed} \cdot 1/V(z_n)$$

where: A_{sed} = surface area of the bottom or sides in contact with the layer

$V(z_n)$ = volume of the layer

Table 9. NOTATIONS AND UNITS USED ALONG WITH VALUES FOR CONSTANTS FOR THE PHOSPHORUS CYCLING MODEL.

Notation	Definition	Dimensions	Value
X1	PHYTOPLANKTON BIOMASS	µg-chl a/l	-
X2	PHOSPHORUS CONCENTRATION	µg-P/l	-
X3	ZOOPLANKTON BIOMASS	µg-C/l	-
X4	DETRITUS MASS	µg-P/l	-
X5	TOTAL PHOSPHORUS	µg-P/l	X2+X4
RO	INCIDENT SOLAR RADIATION	$cal/cm^2/sec$	-
R	AVAILABLE RADIATION	$cal/cm^2/sec$	-
T	TEMPERATURE	°C	-
H	MIXING DEPTH	meter	-
k1	PHYTOPLANKTON MAXIMUM UPTAKE RATE	day^{-1}	0.22*
k2	PHOSPHORUS HALF SATURATION CONSTANT	µg-P/l	0.62*
k3	PHYTOPLANKTON RESPIRATION RATE	day^{-1} $°C^{-1}$	0.0005
k4	ZOOPLANKTON MAXIMUM GRAZING RATE	µg-chl a/l/ µg-C/l/day	0.015*
k5	PHYTOPLANKTON SINKING RATE	$m\text{-}day^{-1}$	0.2
k6	PHOSPHORUS CONCENTRATION OF INFLOW	µg-P/l	-
k7	DETRITUS REGENERATION RATE	day^{-1}	0.035*
k8	ZOOPLANKTON DEATH RATE	day^{-1}	0.3
k9	chl a HALF SATURATION CONSTANT	µg-Chl a/l	1.0*
k10	PHYTOPLANKTON DEATH RATE	day^{-1}	0.05
k11	SETTLING VELOCITY OF DETRITUS	$m\text{-}day^{-1}$	0.1*
k12	NET SEDIMENT RELEASE RATE (ANAEROBIC)	$µg\text{-}P/m^2/day$	2300.0*
ZRES	ZOOPLANKTON MAXIMUM RESPIRATION RATE	day^{-1} $°C^{-1}$	0.0015
CO	PHOSPHORUS TO CARBON RATIO	mg-P/mg-C	0.02
CO1	CARBON TO chl a RATIO	mg-C/mg-chl a	50.0
CO2	FRACTION OF ZOOPLANKTON DIGESTION	mg-C/mg-C	0.5

*Values determined either in situ or in laboratory experiments for Lake Sammamish.

SECTION 8
RESULTS

EXTERNAL NUTRIENT LOADING

Rock (1974) reviewed problems in calculating the pre-diversion nutrient loading in Issaquah Creek, particularly dealing with the choice of either calendar or water year data. By weighting the data on a monthly time basis the short lived high peaks in P content and flow that would bias the annual input estimate were damped out. Thus, while several different approaches (by different individuals) to calculating nutrient load had produced values ranging from 23,000 to 11,000 Kg P/yr, Rock established that about 16,000 Kg P/yr was the most reasonable estimate for WY 1965 and 12,000 Kg P for the calendar year. Total N input was 170,000 Kg N for WY 1965; 122,000 of that was inorganic N. These were determined by multiplying mean monthly flows by mean monthly concentrations.

Post-diversion loading estimates in Issaquah Creek showed considerable year-to-year variation. Moon estimated 8,500 Kg P and 100,700 Kg N (inorganic) in Issaquah Creek for WY 1971. Guttormsen (1974) estimated 198,000 Kg inorganic N in 1972-73 and 258,000 total N. Rock calculated 15,000 Kg P from the 30 samples collected in WY 72 and 7,126 Kg P for WY 73 when a complete data base on P concentration was available. Much of the year-to-year variation was due to flow since the mean annual flow in WY 1972 was 5.58 m^3/sec and 2.91 m^3/sec in WY 1973. The input from June 1974 to June 1975 was 7,128 Kg P, almost identical to the 1973 WY value even though the mean flow was higher--3.98 M^3/sec.

Rock found that most monthly or twice monthly sampling procedures would result in underestimates of the true Issaquah Creek load by 13 to 26%. However, if the 10 year annual flow and the annual mean concentration of 65 µg/l P (12-312 µg/l) was used there was almost perfect agreement with the true load. The difficulty with estimating annual nutrient input involves the simultaneously large, but short lived increases in

flow and concentration. Missing such important points will lead to greatly underestimated loads. For example, 25% of the annual P load came into Sammamish in the 1973 WY during two days in December and two days in January. The mean flow for those four days was 22.6 m^3/sec and the mean concentration was 230 µg/l P. In 1974-75 there were four similar days in January with a mean flow of 20.7 m^3/sec and a mean concentration of 303 µg/l P. Those four days contributed 30% of the 1974-75 load. Thus, it is clear that daily observations of concentration as well as flow are necessary, particularly during periods of high flow, if serious underestimates of loading are to be avoided.

Thus, it was important to attempt to normalize flow during pre- and post-diversion years to obtain an accurate estimate of the quantity of P diverted. For flows below 2.8 m^3/sec., flow and P concentration in 1973 were inversely correlated, but above that flow they were directly correlated. Both correlations were statistically significant. The process operating that seems to produce such variation is a seasonal one. As the flows increase in the fall and winter the material accumulated in the streams and on the land during the summer is flushed out. During that period concentration may increase dramatically with flow. Once the material has been flushed out in late winter and spring, increased flow may carry very little particulate matter with it and concentrations may decrease in spite of high flows. For example, for flows over 2.8 m^3/sec the November-December-January mean was 88 µg/l while the February-June mean was 39 µg/l. The annual mean was 65 µg/l.

To accurately estimate the fraction of P diverted, years of equal flow must be compared. The 1973 WY was a very low flow year compared to the 1965 WY, however, when all post-diversion data were used the resulting correlation showed a relatively good fit ($r = 0.41$) and it seemed reasonable to let the equation describe an average post-diversion year. The mean flows for all post-diversion data through the 1973 WY is 4.3 m^3/sec., essentially the same as the mean pre-diversion flow of 4.25 m^3/sec.

The estimated P loading calculated by the above post-diversion equation is 9000 Kg. Thus, comparison of this value with the most reasonable pre-diversion estimate of 16,000 Kg P leaves 7000 Kg P diverted by the METRO project in 1968. Assuming Issaquah Creek is now 70% (Moon, 1972) of the surface income to the lake this means that about 35% of the total surface input P was diverted. Including an estimate for atmospheric input to the lake surface lowers the diverted percentage to 34. If around 7200 Kg P is considered as the post-diversion Issaquah Creek input, because of the close agreement in 1973 and 1974, then the diversion would be 8800 Kg P or 43%, which is probably high.

Another approach was used to estimate the fraction diverted and that was to back calculate the quantity of waste-water P that was probably entering before diversion. As part of this, METRO conducted a mass balance of the inputs and outputs to and from the Creek within the town of Issaquah during high flow and low flow. In addition the Issaquah treatment plant flow before diversion was used with literature values of dairy waste and sewage P to estimate the wastewater P load before diversion (see Rock, 1974,for details). The P loads measured in the Creek did not balance and thus failed to provide usable data because of diurnal variations, but the indirect approach provided an estimate of 4,400 Kg P diverted, which would have been 33% of the Issaquah load (normal flow year), but only 25% of the lake's total load. Using this approach would mean that Issaquah Creek carried a pre-diversion load of 13,400 Kg P.

Because of the many uncertainties in the latter approach it seemed most reasonable to stick with the previously mentioned approach; that of correcting the post-diversion data to a normal flow year, which the 1965 WY apparently was,comparing that to the 16,000 Kg best estimate for Issaquah Creek before diversion. Therefore, the most probable estimates of pre- and post-diversion phosphorus loads are shown in Table 10. The remaining wastewater P input comes from the salmon hatchery in Issaquah, which is a rather crude estimate based on the two mass balances done on the various inputs to Issaquah Creek by METRO.

Table 10. PHOSPHORUS LOADING (Kg/Yr)TO LAKE SAMMAMISH BEFORE AND AFTER WASTEWATER DIVERSION. P ESTIMATES ARE BASED ON A NORMAL WATER YEAR.

	Pre-diversion (%)	Post-diversion (%)
Issaquah Creek		
sewage and dairy waste	7,000 (34)	0 (0)
land runoff	8,500 (42)	8,500 (64)
salmon hatchery	500 (3)	500 (4)
11 Minor Tributaries	3,900 (19)	3,900 (29)
Atmosphere to Lake Surf.	400 (2)	400 (3)
Total external loading	20,300 (100)	13,300 (100)
Loading in g/m^2 yr	1.02	0.67

In terms of normal year-to-year variation it was noted that for the 1972 WY, which was wet, the Issaquah Creek loading was 15,000 Kg P. For the 1973 WY, which was dry, the loading was 7,110 Kg P, with 9000 Kg P as the estimate for an average or normal post-diversion year. The respective flow weighted mean concentrations for the wet 1972 and dry 1973 WY's were 85 and 78 µg/l. The surprising thing is that 1974-75 flow was about normal, 3.98 m^3/sec versus the 11-year mean of 4.25 m^3/sec, and the flow-weighted mean concentration was 57 µg/l. Thus, there may be more variation in flow from year to year than in concentration.

If these post-diversion flow-weighted mean concentrations for 1973-75 are compared with the pre-diversion concentration (68 versus 119 g/l) then that represents a 43% reduction in concentration in Issaquah Creek or 34% for all inflows if Issaquah Creek is 70% of the P input. That estimated reduction in concentration is in agreement with the reduction in loading.

Nevertheless, the P load can vary annually due largely to flow on an order similar to the fraction directed. This again emphasizes the necessity of comparing relatively similar hydrologic years to estimate manipulated loading changes.

The nitrogen balance was treated less intensively than phosphorus because the lake is primarily P limited and the difference in analytical methods for NO_3 and organic N make the comparison of before and after diversion data rather questionable. However, the various estimates of the total N input for two post-diversion years compared to METRO's estimate of pre-diversion input (NO_3 corrected to results with the Cd-Cu reduction method - see Emery, 1972) are shown in Table 11. To extrapolate to total lake input, Issaquah Creek loading was assumed to be 57% of the total surface water input. See Appendix B for 1974 N loading to Issaquah Creek--it was nearly the same as that observed in 1972-73, the basis for Table 11.

Table 11. NITROGEN LOADING TO LAKE SAMMAMISH IN Kg/Yr; 11 TRIBUTARIES PLUS ISSAQUAH CREEK.

	Pre-diversion (1965 WY)	Post-diversion (3/72 - 2/73; Guttormsen, 1974	Post-diversion (6/74 - 6/75
Organ NH_3-N	69,000	60,000	135,600
$NO_2 + NO_3 - N$	174,000	198,000	179,470
Total N	243,000	258,000	314,730
Loading in $g/m^2 \cdot yr$	12.3	13.0	15.9

The increase noted in Table 11 cannot be considered significant in the face of the analytical differences between pre- and post-diversion period investigators. However, the ratio of N:P loading is 12 before

and 20 after diversion, suggesting that P should be most limiting under most circumstances. Nitrogen fixation has been shown to be rather insignificant to the total N supply (J. Staley, personal communication[1]). Atmospheric input has not been measured, but should not account for more than a few percent of surface water input.

RESPONSE TO DIVERSION

Common trophic state indicators in Lake Sammamish have remained relatively stable, or at least no significant trend is apparent, in the six post-diversion years the lake has been monitored. Figure 5 shows the mean concentrations of total P and NO_3-N measured at station 612 during December through February (the non-growth months) when the lake is well mixed. These values are weighted means in the photic zone, which corresponds reasonably well with the epilimnetic depth in this lake, and are compared with the pre-diversion values assigned as 100%. P remained very stable, well within 10% of the pre-diversion level. Although NO_3 decreased, the correction of prediversion data for a methods difference (a factor of 2) makes the significance of that decrease rather doubtful. In any event levels are very near 100% in the last two years.

Chl *a* concentration was determined from March through August and weighted for time, similar to the analysis of stream nutrient data mentioned earlier, so that months with more frequent data points would not be weighted disproportionately heavy. Although chl *a* has varied considerably from year to year no trend of either increasing or decreasing algal biomass in the lake is apparent. The cause for such variations in spite of stable pregrowth P levels is not related to year-to-year variation in inflow P, but probably to internal mechanisms of P supply as will be discussed in later sections. In association with algal biomass, Secchi disk depth measurements have also not changed. The mean summer season Secchi measurement remains at 3.3 m, in fact the greatest Secchi disk value recorded is 6.1 m - a pre-diversion value.

[1] J. Staley, Dept. of Microbiology, Univ. of Washington, Seattle.

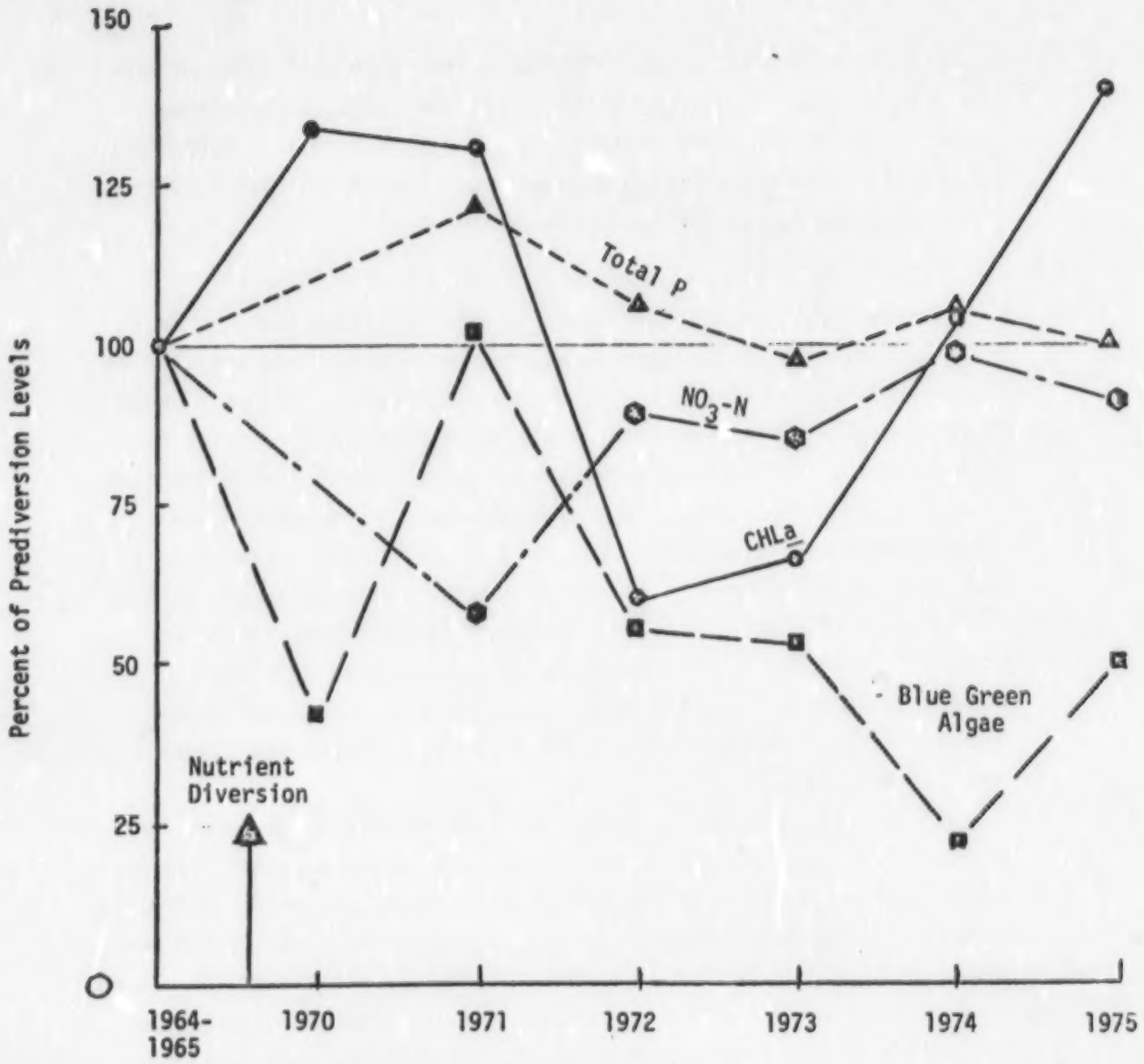


Figure 5. Mean concentrations in the photic zone (usually top 8 m) of growing season chl a (Mar-Aug) and winter (Dec-Feb) total phosphorus and nitrate nitrogen relative to pre-diversion 1965 levels. The 1965 levels were: chl a 6.5 µg/ℓ (actually a mean of 1964 and 1 65 data), total P 31 µg/ℓ and NO_3-N 390 µg/ℓ. The % blue green algae of the total phytoplankton volume was compared against the pre-diversion mean for June-October in 1965 and July-Oct in 1964 (67.5%).

Oxygen deficit rate has remained right at 0.05 mg $O_2/cm^2 \cdot day$ since diversion, a little more than the pre-diversion rate, but with year-to-year variation no more than a few $\mu g/cm^2 \cdot day$.

The one variable that seems to have changed significantly is the fraction of the growing season (in this case June through October) phytoplankton that is composed of blue green algae. This value, determined from surface and/or 1 m depth samples, has been consistently below that of the 1964-65 pre-diversion period, and except for one year has been markedly less. Including all years the average is 46% below the pre-diversion level.

The lake can clearly be considered as mesotrophic with respect to algal biomass and productivity. However, nutrient content and oxygen deficit would suggest a lake at least on the border between mesotrophy and eutrophy (Table 12). From the standpoint of Carlson's (1974)[1] technique it falls into the 40-50 numerical range based on total P, chl *a* and Secchi disk depth.

Table 12. SUMMARY OF PRESENT LIMNOLOGICAL CHARACTERISTICS IN THE PHOTIC ZONE (EXCEPT OCYGEN DEFICIT).

Parameter	Annual Mean	Growing Season* or Winter Means**
Chl *a*	4.0 mg/m^3	6.0 mg/m^3
Primary Production	494 mg $C/m^2/d$	700 mg $C/m^2/d$*
Total P	24.4 µg P/l	31 µg P/l**
Ortho P (photic zone)	7.9 µg P/l	12 µg P/l**
NO_3+NO_2-N	180 µg N/l	275 µg P/l**
Oxygen Deficit (15-30 m)	0.049 mg O_2/cm^2-day	

*Growing season is March to August

**Winter is December to February

[1]Carlson, R., McGill Univ., Montreal, Canada, unpublished manuscript.

WATER COLUMN PHOSPHORUS

The cycle of P in the epilimnion of Lake Sammamish is somewhat variable from year to year, but generally concentrations of both total and ortho P are maximized at water column turnover during November - December (Fig. 6). Levels usually less than the maximum persist during winter (Jan. - Feb.). Ortho P is depleted to a slightly greater extent than total P in spring and summer due to the conversion of P into algal biomass. The concentrations of ortho P reached near undetectable levels in the summer during some years, e.g. 1974.

The post-diversion annual mean epilimnetic content of total P is about 24 µg/l and ortho P 8 µg/l. The winter (Dec. - Feb.) mean for total P is 31 µg/l and 12 µg/l for ortho P. One can see from Fig. 6 that 1974 represents a rather typical year with respect to mean values of the two forms of P. The annual mean total P content for the entire water column, which includes the anaerobic hypolimnion in summer, is 36 µg/l.

The question of the stable 30 µg/l winter concentration in Lake Sammamish in spite of changed P loading is of extreme interest and has been a major thrust of this research. Intensive monitoring (weekly) of the turnover period during October through December was undertaken by Rock (1974) to approach the problem.

The lake is stratified from mid May to mid November. Portions of the hypolimnion are anoxic from late July or early August until turnover (Fig. 7).

The pattern of D.O., total P and total Fe are shown in Figures 8 and 9 during 1972 and 1973. Note the inverse relationship between D.O. and Fe and D.O. and total P. Although concentrations in the hypolimnion seem potentially capable of at least doubling the epilimnetic concentration at overturn it does not happen presumably because once the hypolimnion becomes aerated and Fe is oxidized much of the hypolimnetic P is resedimented. The high levels of Fe and P were not observed in 1973 as

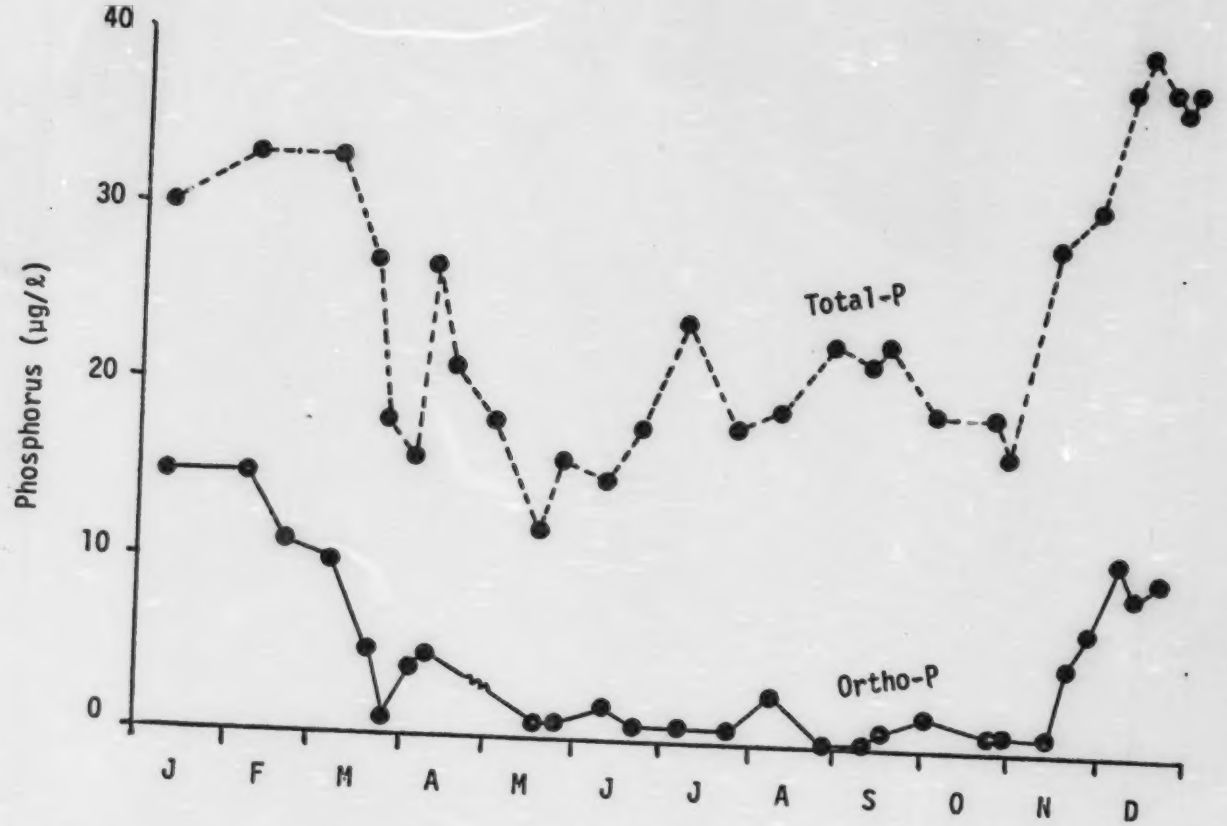


Figure 6. Total and ortho phosphorus concentration (mean of epilimnion) at Lake Sammamish central station in 1974.

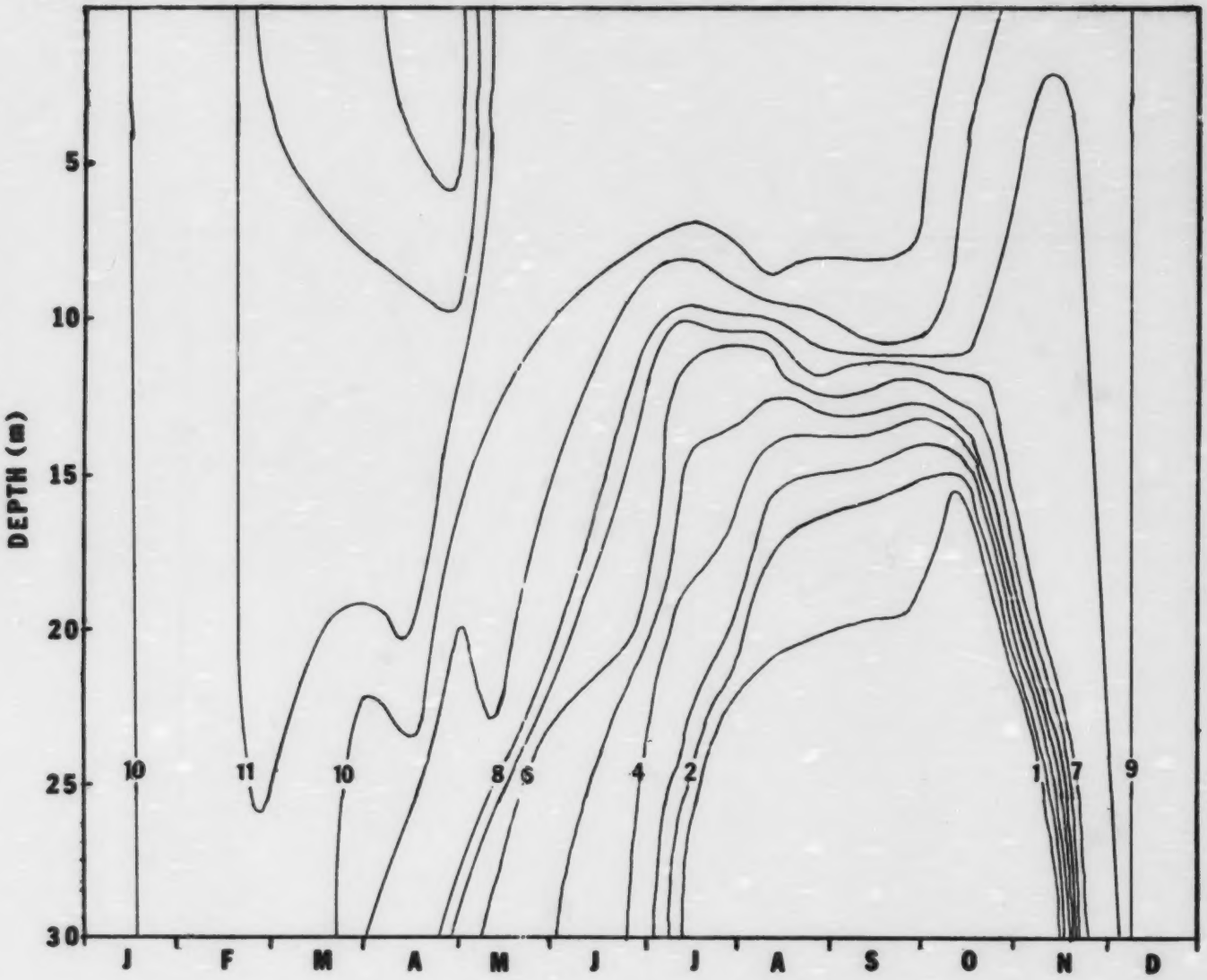


Figure 7. Oxygen isopleths in Lake Sammamish for 1973.

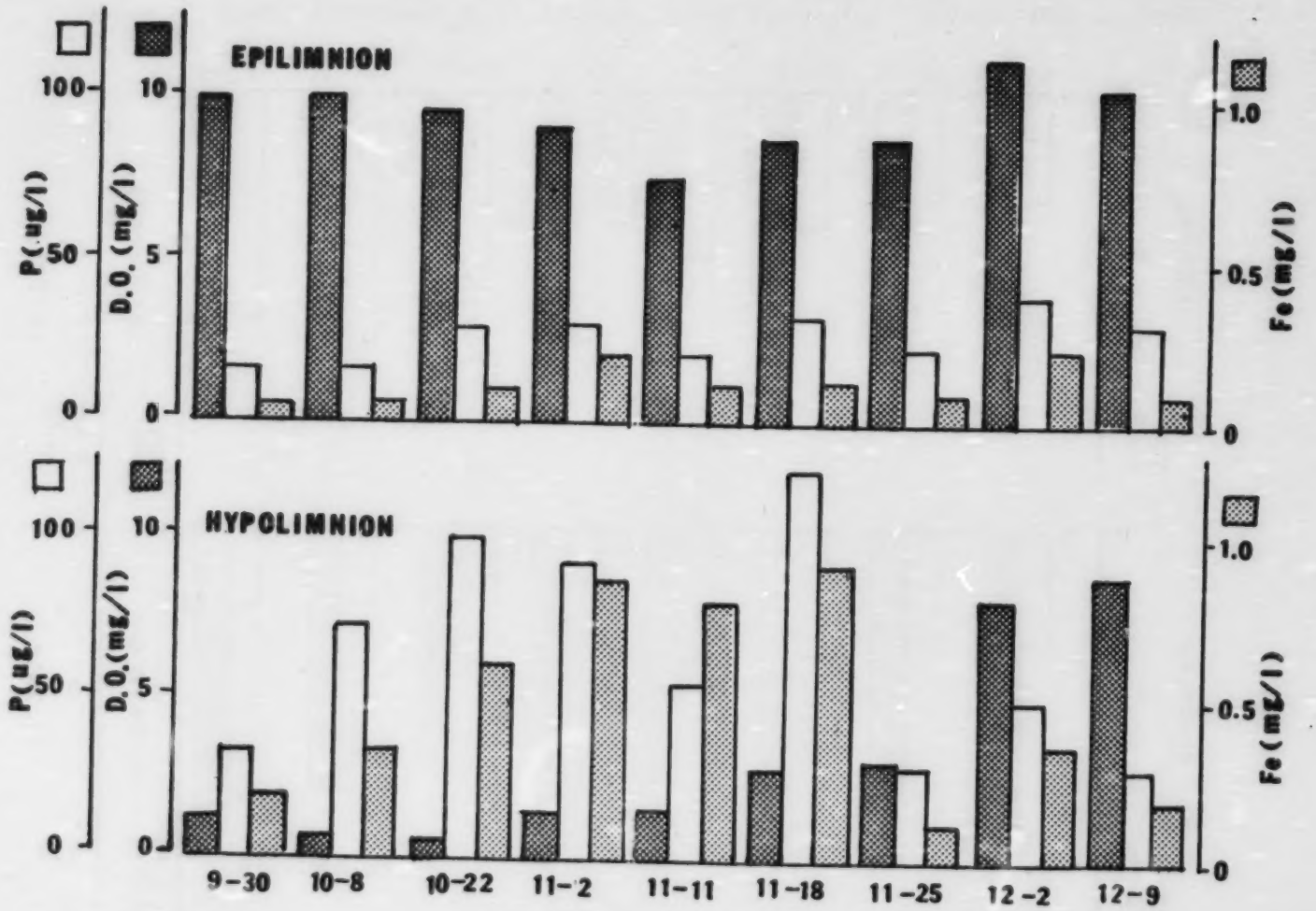


Figure 8. Dissolved oxygen, total phosphorus, and total iron concentrations in Lake Sammamish for the 1972 turnover period (from Rock, 1974)

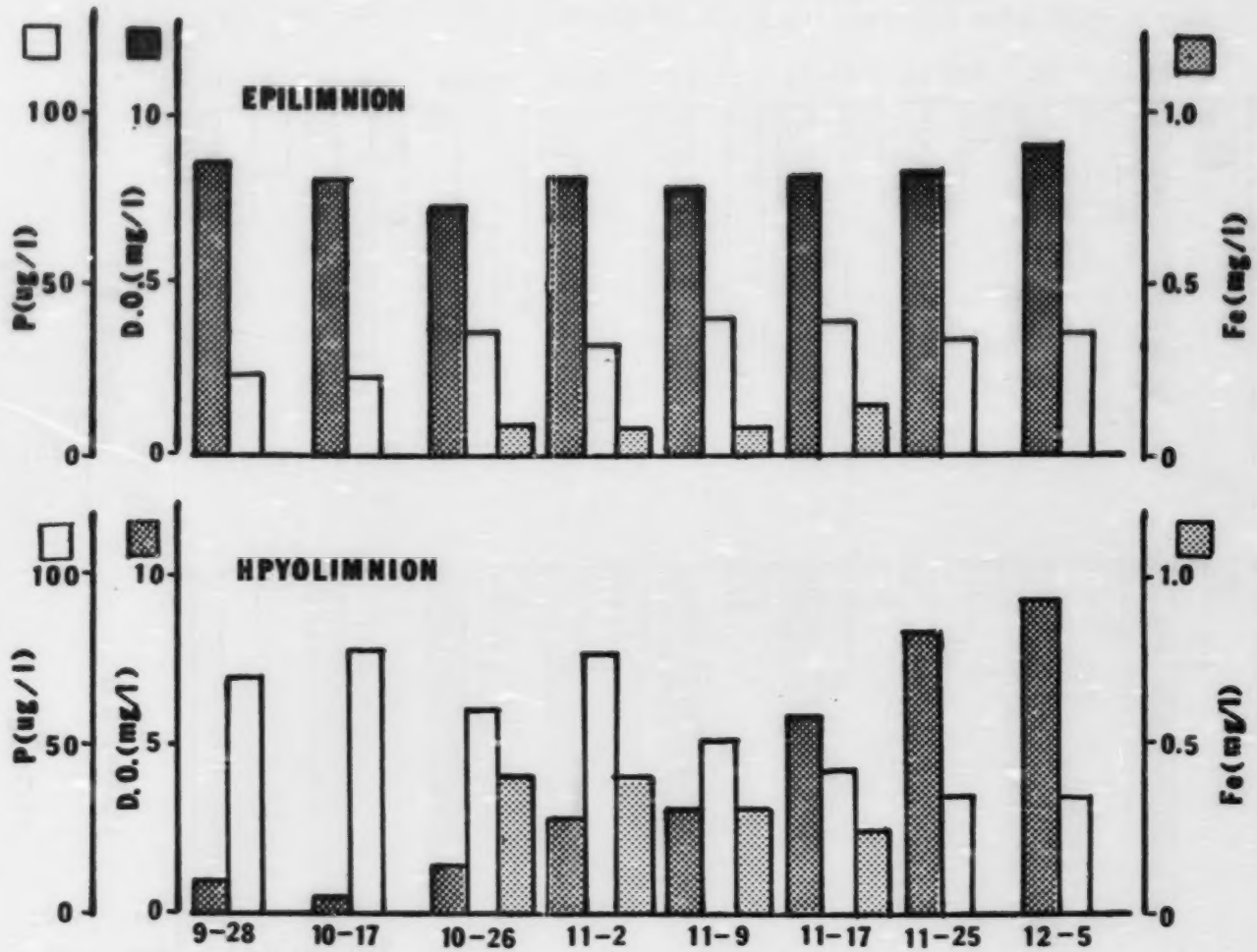


Figure 9. Dissolved oxygen, total phosphorus, and total iron concentrations in Lake Sammamish for the 1973 turnover period (from Rock, 1974)

in 1972 because the D.O. content began to increase earlier (compare early November values) than in 1972. Thus, earlier aeration of the hypolimnion initiated earlier oxidation and sedimentation rate increase in 1973.

This efficient sedimentation of hypolimnetic P as a result of aerating a previously anaerobic water mass may be the stabilizing force for the water column P concentration. Thus, added P to the lake, particularly if added to the hypolimnion during the warm stratified period, could simply be removed through the sedimentation process and never be available for spring-summer algal growth. In fact, the inflow from Issaquah Creek probably does not enter the epilimnion, but rather dives to the lower part of the metalimnion as shown by a dye study (Moon, 1972).

There has been an apparent change in the peak water column P content after diversion, that is, the total amount of P in g/m^2. Figure 10 shows how these peak values have substantially decreased in the post-diversion years 1972-73. Krull (1973) suggested that the high pre-diversion peaks were related to the rate of water column breakup at turnover, but Rock (1974) showed that was probably not the total cause for the differences in Figure 9 because turnover or temperature decline rates for 1964, 1965 and 1973 were similar. However, he did hypothesize that the difference in calculated P retention before (0.848 $g/m^2 \cdot yr$) and after (0.489 $g/m^2 \cdot yr$) diversion was the result of sedimentation differences, because all other evidence points to the release rate from sediment remaining constant. He argued that the excess pre-diversion sedimentation could have resulted in a build-up of loose particulate matter on the lake bottom and thereby was supplying the amount of P necessary to explain the higher pre-diversion peaks (Fig.10) through mixing.

PHYTOPLANKTON

Phytoplankton biomass shows a peak in the spring composed primarily of diatoms. The dominating genera during winter and spring are Melosira and Stephanodisous. During the summer and fall, Fragilaria, Synedra,

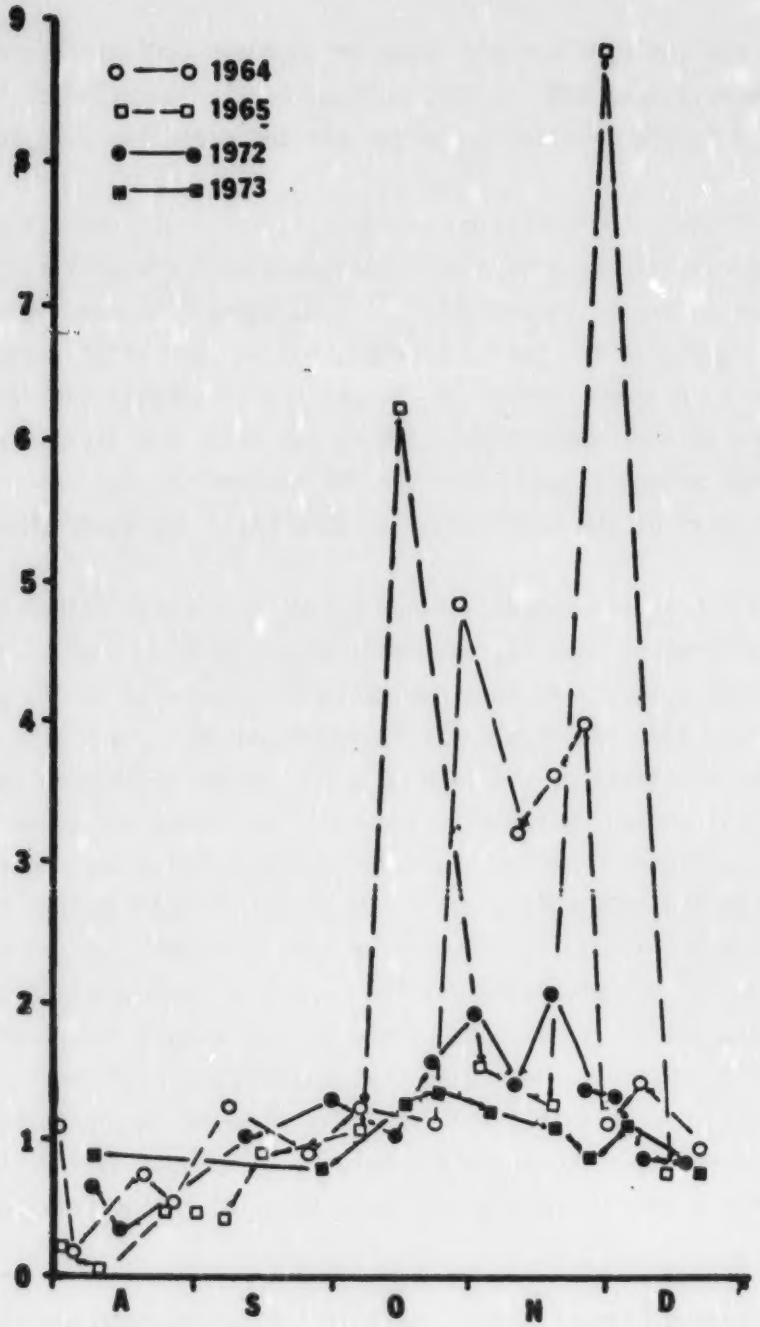


Figure 10. Water column phosphorus content for turnover period (after Krull, 1973)

Melosira, Rhizosalenia and Asterionella are the major diatoms. The blue-green algae are comprised predominantly of Aphanocapsa, Microcystis, Coelosphaerium, Anabaena and Gomphosphaerium. In 1973-74 the appearance of Aphanizomenon has been less pronounced than earlier while the abundance of Lyngbya has increased. Predominant chlorophyseans are Oocystis, Sphaerocystis, Closteriopsis, Chlamydomonas and Staurastrum. Also predominant in the phytoplankton of the lake is the chrysomonad Mallomonas.

The pattern of phytoplankton biomass is one of typically the largest concentrations occurring in the spring - the diatoms (Fig. 11). During some years biomass increase begins early (March) while in other years the increase is delayed until April. However, in all years the chl a content has subsequently declined to 5 µg/l or less. Thus, although the composition of blue green algae comprised from 15 to 69% of the June - October biomass in post-diversion years (1970-75) the abundance of blue greens is not great.

The maximum chl a content shown in Fig. 11 was 28 µg/l in 1971. That equals the pre-diversion maximum in 1965. However, the highest value was in 1975 - 37 µg/l in mid April.

The growing season mean chl a content is 7 µg/l and that value has remained rather stable during the 6 year study. However, growing season productivity can vary as much as ± 100% from year to year (Table 13). Curiously, the years with low mean productivity are also the years when the diatom bloom has started in early March. Productivity was not measured by Metro by similar methods so comparison of normalized (for

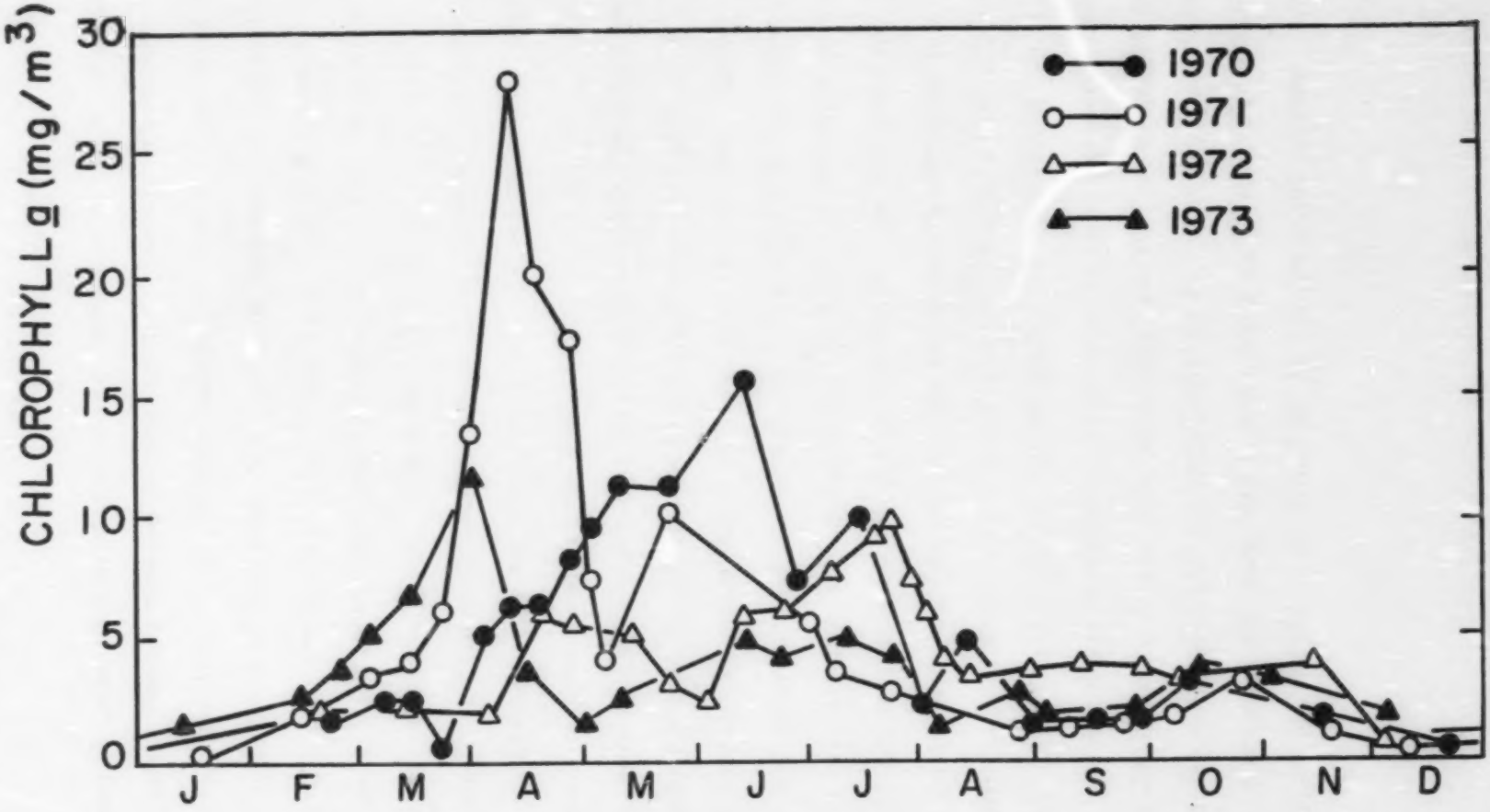


Figure 11. Temporal variation in chlorophyll a (integrated means for photic zone) in Lake Sammamish, 1970 through 1973 (1970-71 data from Emery 1972)

Table 13. ANNUAL AND GROWING SEASON MEANS OF PHYTOPLANKTON CHLOROPHYLL a (WEIGHTED MEANS FOR THE EUPHOTIC ZONE) AND DAILY RATE OF PRIMARY PRODUCTIVITY IN 1970-1974.

Lake Sammamish

Year	Chlorophyll a (mg/m^3)		Primary Productivity ($mg\ C/m^2 \cdot day$)	
	Yearly	Growing Season	Yearly	Growing Season
1970	5.7	7.7	711	899
1971	6.6	10.9	467	575
1972	4.3	4.8	799	952
1973	4.0	4.7	496	545
1974	6.0	6.8	789	904
Average	5.3	7.0	652	775

light)values before and after diversion is not possible.

Transparency of the water column has been even more stable and less altered after diversion than chl a. As Fig. 12 shows the mean summer value has remained very constant around 3.5 ± 0.2 m. Interestingly enough an August value taken in 1913 by Kemmerer, et al. (1923) showed 3.3 m.

ZOOPLANKTON

Vertical net hauls were collected at station 612 (see Fig. 3) at frequencies varying from twice weekly to once per month with the least frequency at periods of low reproductive activity. Relatively little zooplankton work had been done in Lake Sammamish prior to 1970. The first recorded zooplankton haul in Lake Sammamish was made by Kemmerer, et al. (1923) in 1913. Table 14 shows the results of this vertical net haul (18 m) from August 13, 1913, versus the results from a net haul (25 m) made on August 14, 1972.

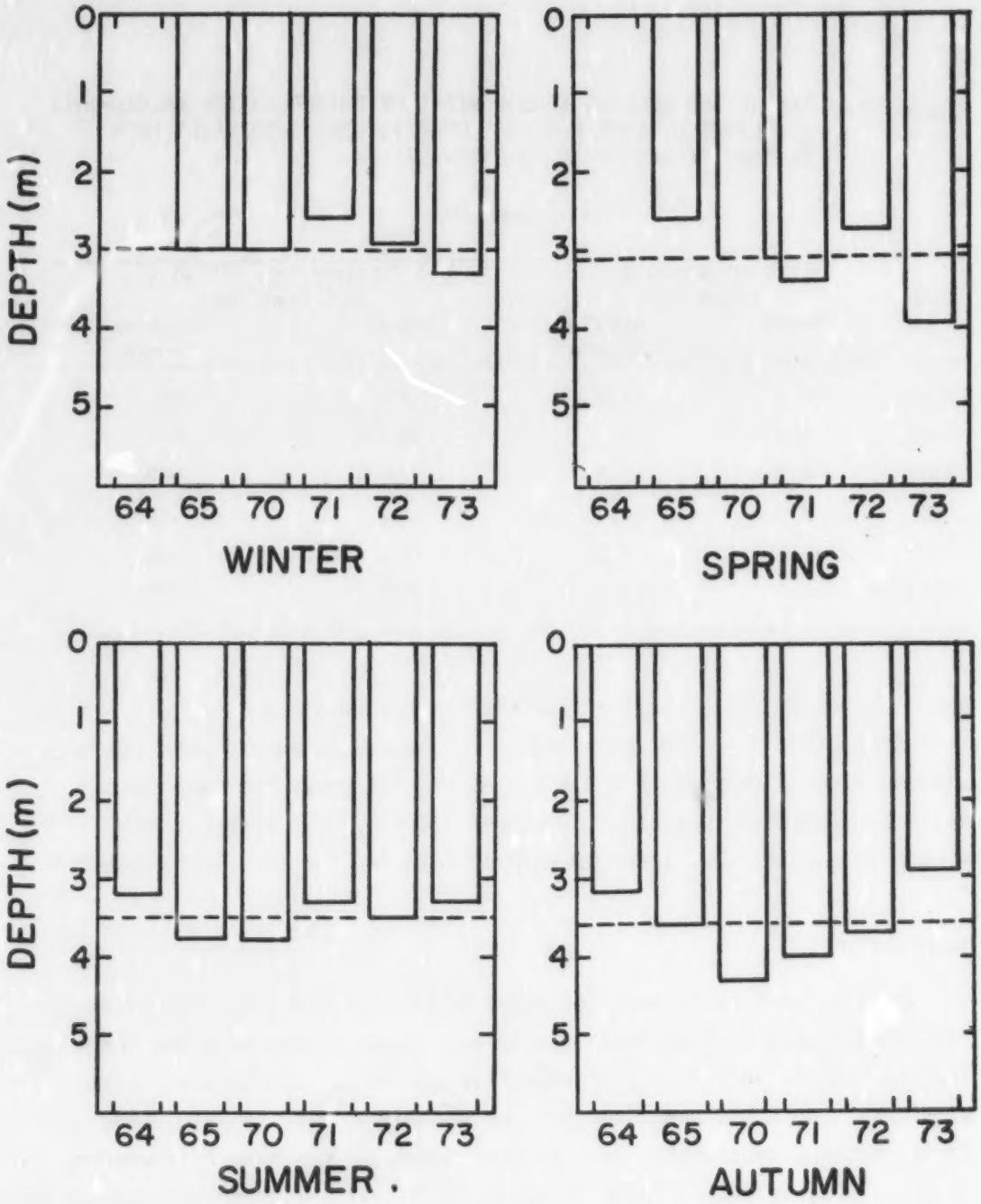


Figure 12. Mean seasonal variation in Secchi disk readings for Lake Sammamish. Dashed line represents mean for the plotted data (1970-71 data from Emery, 1972)

Table 14. COMPARISON OF AUGUST NET HAULS 1913 vs 1972 (NUMBER PER m^3).

1913			1972	
		Cladocerans		
Bosmina	393		Bosmina	51
Diaphanasoma	905		Diaphanasoma	331
		Copepods		
Cyclops	5780		Cyclops	3268
Epischura	9805		Epischura	151
Diaptomus	-		Diaptomus	6573
		Rotifers		
Mastigocerea	393			0
Notholca (=Kellicottia)	113			0
Polyarthra	113			0

The two samples are quite similar except for Diaptomus being absent in the sample collected in Lake Sammamish in 1913, and it is the most common zooplankter now in the lake. The difference in numbers in the other species with only 1 sample as a comparison is not significant as population peaks may be shifted several weeks from year to year.

The first major zooplankton study of Lake Sammamish was a portion of the METRO study of June, 1964 to December, 1965 (Isaac, et al., 1966). Figure 13 is a graph of total zooplankton (number per m^3) in 1965 versus 1972. The total numbers are quite similar and any difference probably can be explained on year-to-year or sampling variation. For example, the major peak in both years was due to a rotifer bloom, which reached 160,000/m^3 in 1965 and only about 90,000/m^3 in 1972. This could be explained by postulating that the 1965 sample was taken at the height of the bloom while the 1972 sample was a week on either side of the peak. Species recorded during the 64-65 study are the same as those found in the lake at the present time.

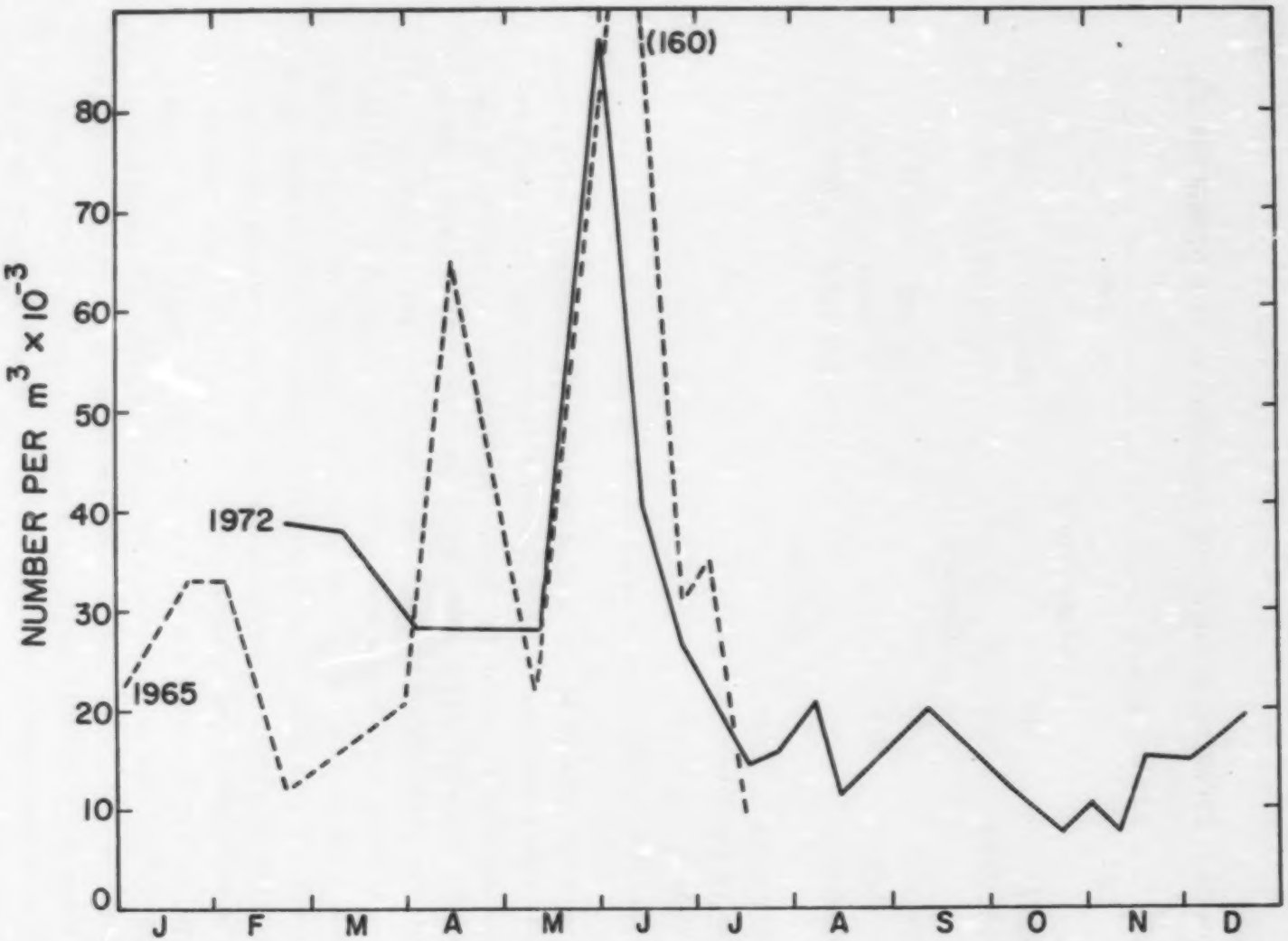


Figure 13. Total zooplankton in Lake Sammamish, 1965 vs 1972

Table 15 is a list of the zooplankton species that have been collected in Lake Sammamish during this study. Figure 14 shows the per cent numerical occurrence of the three major groups of zooplankton throughout the year and the most abundant is the copepod Diaptomus ashlandi.

Table 15. ZOOPLANKTON SPECIES IN LAKE SAMMAMISH.

Copepods	Rotifers
*Diaptomus ashlandi	*Kellicottia longispina
*Epischura nevadensis	K. bostoniensis
*Cyclops bicuspidatus	*Polyarthra sp.
	Keratella cochlearis
Cladocerans	K. Quadrata
*Daphnia thorata	*Conochilus unicornis
*D. schølderi	Collotheca mutabilis
*Bosmina longirostris	C. pelagica
*Diaphanasoma leuchtenbergianum	Notholca squamula
Leptodora kindtii	Ploesoma hudsoni
Scapholeberis kingi	Gastropus sp.
	Synchaeta sp.
	Trichocerca sp.
	Filinia sp.

*indicates most common species.

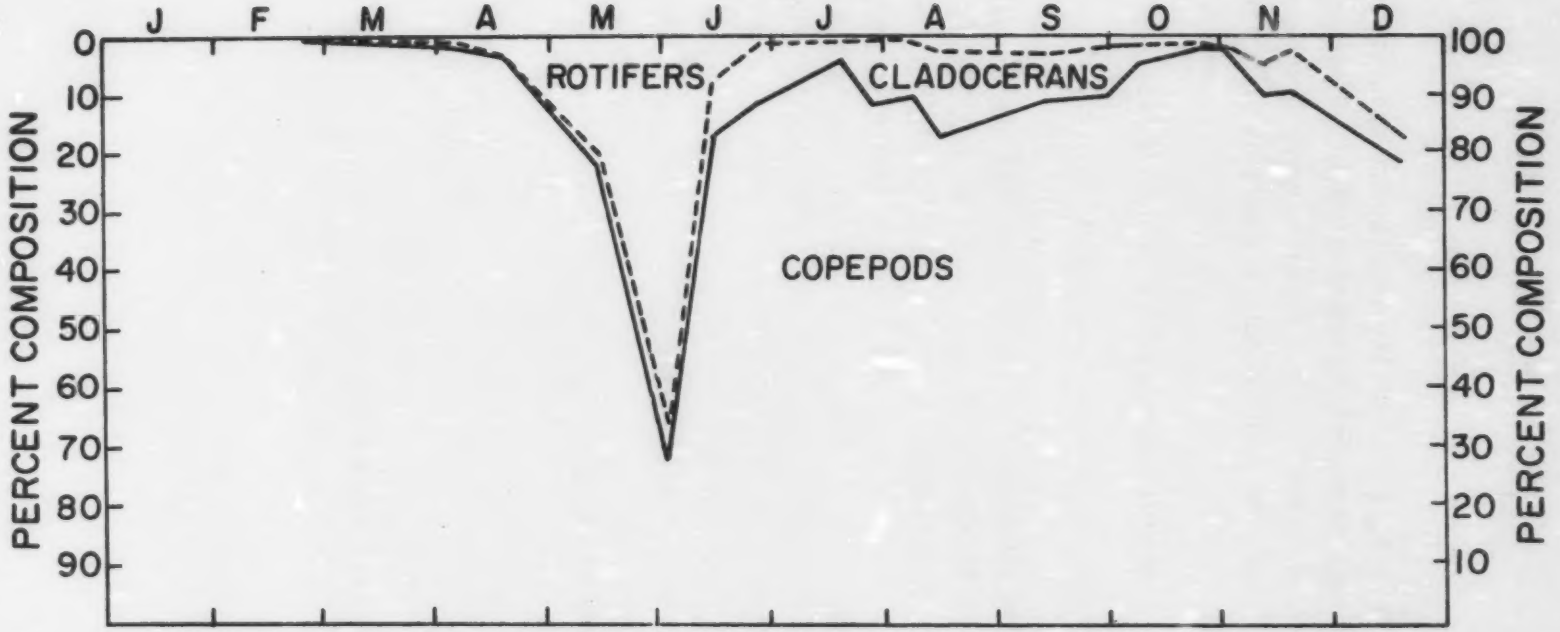


Figure 14. Percent composition of zooplankton categories, 1972

SEDIMENTATION

Sediment trap data are summarized in Table 16 while seasonal events are illustrated in Figures 14 and 15. The trend for increasing sedimentation with depth was due to relatively coarse and dense allochthonous material transported into the hypolimnion which settled rapidly and tended to be intercepted by deeper traps. Also in late autumn sediment released P continuously coprecipitates with Fe-hydroxy flocs as the thermocline erodes and upper hypolimnetic waters are reaerated. The Fe floc-P material was collected by deeper traps although at overturn some of it was also collected by shallow traps. The littoral (Station 3) and sub-littoral (Station 1) traps collected only about 70% as much as traps suspended at the same depths in pelagic regions (Stations 2a and 612) mainly because they were somewhat sheltered from alochthonous inputs during times of high stream flow.

With regard to seasonal events (Figures 15 and 16), high sedimentation rates occur during spring, late autumn and winter. Microscopic examination of freshly collected material indicated that high sedimentation rates in the spring resulted from settling out of the diatom bloom which occurred in March-April. During summer the settled material was dominated by blue-green algae detritus. In situ experiments (Birch, 1976) indicated that this material decomposes much more rapidly than diatom detritus, a factor believed responsible for the relatively small collections in traps at that time.

In late summer an unusual organism tentatively identified as a bacterium Metallogenium Personatum appeared in abundance in the sample from 26 m and in the autumn in the 15 m samples as well. This organism has also been found in hypolimnotic water samples from Lake Washington (Shapiro et al., 1971). The radiating tubules of this organism are coated with brown Fe and Mn containing compounds (Libby, 1972) to which P may be adsorbed. The significance of this mechanism in Fe, Mn and P cycling in Lake Sammamish has not yet been clearly identified.

Immediately after thermal destratification (12/4/74) there was a

spectacular increase in the rate of sedimentation due to the chemical effect of reaerated bottom waters. Most of the Fe which had been released following reduction of surface sediments during the anaerobic period was reoxidized and precipitated as a floc. Sediment released P is sorbed on to these Fe (III) hydroxy compounds and co-precipitated. Since sediment released P, until overturn, was contained in lower hypolimnetic waters and below the photic zone its contribution to algal nutrition in Lake Sammamish is considered minimal.

During winter high sedimentation rates of total material and of P were due to increased input of allochthonous material resulting from high stream flows. More material, particularly total sediment, was deposited at Station 2A becuase it is closer to the major inlet, Issaquah Creek. High sedimentation rates of P during winter means that a significant proportion of the external P income, most of which is introduced during winter, is rapidly deposited and effectively removed as a source for spring-summer algal nutrition.

A comparison of primary production and sedimentation into the 7 m trap (Station 612) during the productive season (March-October) indicated that only a very small proportion ($\sim$15%) of this produced material settled out of the photic zone as detritus. This indicates that nutrients are efficiently recycled in the trophogenic zone during the stratified period. The growing season rates of sedimentation were 43, 4.8 and 0.67 mg/m^2 day for C, N and P respectively.

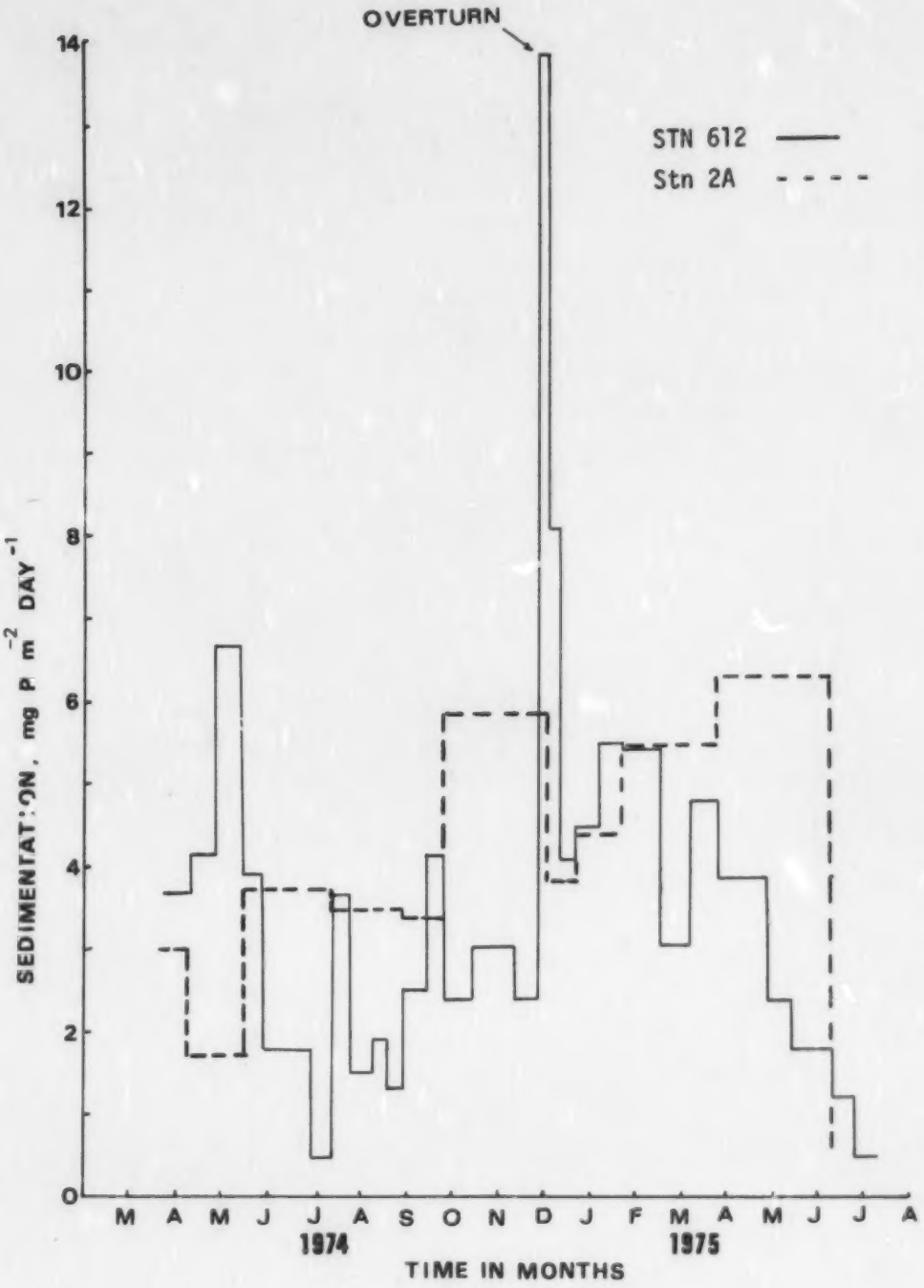


Figure 15. Phosphorus sedimentation rates in Lake Sammamish determined at Stations 612 (23 m depth, 26 M Trap) and 2A (30 m depth, 27 m Trap).

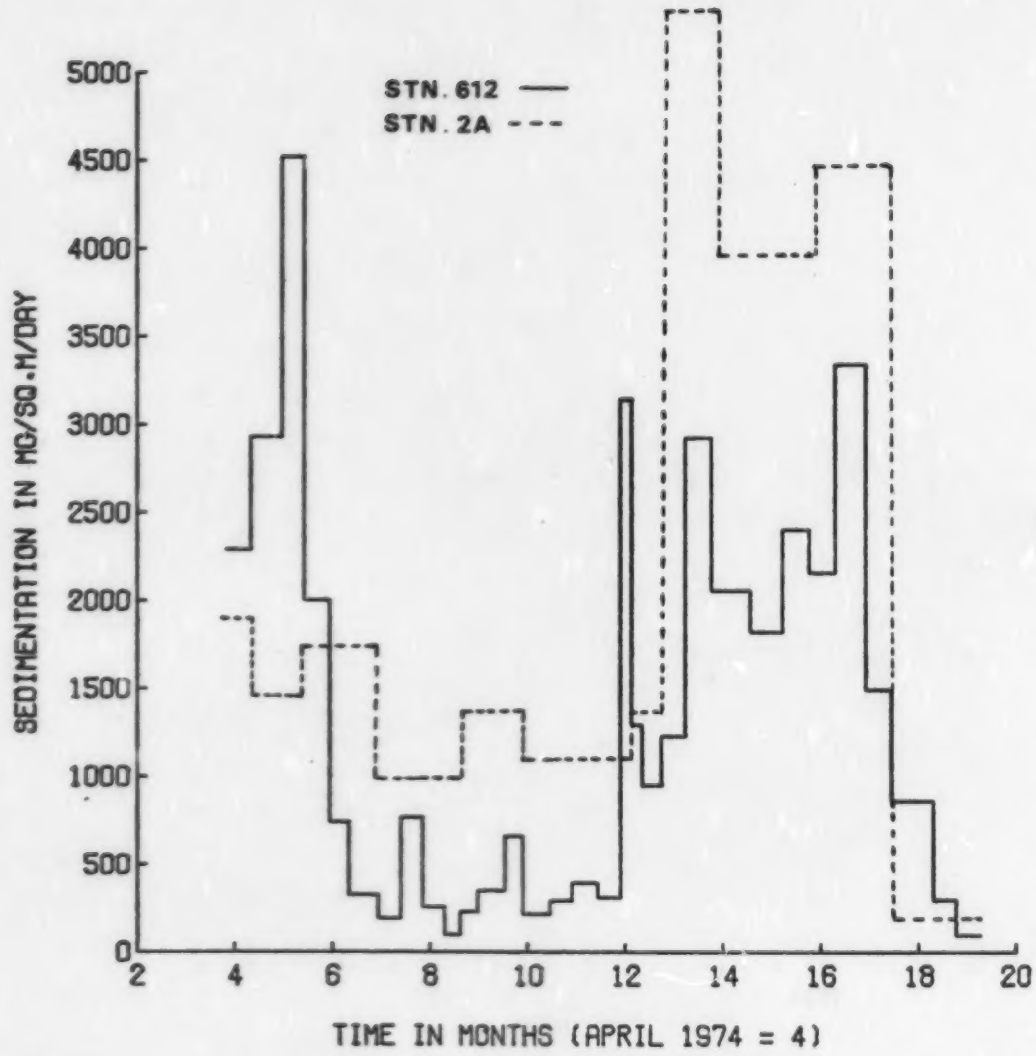


Figure 16. Total sedimentation rates in Lake Sammamish at Station 612 (28 m depth, 26 m Trap) and Station 2A (30 m depth, 27 m Trap).

SEDIMENT-WATER INTERCHANGE OF P

During the course of the column experiments there was a doubling in ambient lake temperature from 8° C in early spring to 20° C in summer and back down to 10° C in autumn. The column DO content decreased rapidly; from 8-11 mg/ℓ to 2-3 mg/ℓ in 30 days, with a subsequent slower decrease to as low as 0.1 mg/ℓ in some columns. DO depletion in transparent columns was much slower due to photosynthesis.

In all cases the loss of DO was accompanied by increases in P and Fe content. Alkalinity and pH responded consistently to the onset of anaerobic conditions. The opaque column's pH decreased to 6.5 and alkalinity increased from 32 to 39 mg/ℓ as $CaCo_3$. Subsequently pH increased to 6.8-7.0. There was an initial increase in pH in the transparent columns due to photosynthesis, but then they attained levels similar to the opaque columns.

Rates of release of Fe and P were determined by considering only positive slopes on the concentration time curves and only after DO had decreased to levels of 2-3 mg/ℓ. Values were excluded where problems of O_2 intrusion were known to have occurred.

As representative results from column experiments Figures 17-19 are presented. Figure 17 shows changes in total P, total Fe and DO in a column initially purged with N_2 to effect early DO depletion. That procedure showed clearly that the time of incubation was indeed short before Fe and P began to release. After day 20 total and soluble Fe increased at rates of 25 and 12 mg/m^2 day, respectively, and total and soluble P also increased at 3.9 and 4.3 mg/m^2 day, respectively.

Figure 18 shows results of an opaque column allowed to lose its DO naturally, which appeared nearly as fast as the previously described experiment in which the column was purged. The rates in this experiment were 3.15 and 3.22 mg/m^2 day for total and soluble P, respectively. Total and soluble Fe increased at 10 and 5 mg/m^2 day. Note the only slight increase in P following the addition of P at 3 times the lake loading rate on day 120.

Although the DO decreased in the transparent column more slowly than in the opaque columns, release rates were as great as in the opaque columns once DO became low (Fig. 19). It took 90 days to reach a concentration of 0.5 mg/ℓ in two such transparent columns, one at 7 m and the other at 10 m depths. Total and soluble P released at 4.4 and 4.1 mg/m^2 day once DO was depleted to the low value and total and soluble Fe at 36 and 32 mg/m^2 day, respectively.

For comparison the observed changes in P and Fe in the hypolimnion are shown in Figures 20 and 21. During 1974, P, Fe and DO were determined at depths of 23-25 and 27 m at Station 612 - the 28 m Station. The results in Fig. 20 and 21 were calculated for a 5 m water column at that deep station. As shown P increased gradually after day 13 (July 8) but not until day 102 (Sept. 10) did the increase appear marked. The rates of release for the period were 5.2 and 4.9 mg/m^2 day for total P and soluble P, respectively, and 25 and 12 mg/m^2 day for total and soluble Fe. The DO loss rate was 0.69 mg/m^2 day which was the same as in one opaque column - the other two opaque columns showed rates of 0.54 and 0.58 mg/m^2 day.

Interestingly enough the maximum P value reached, 290 µg/ℓ (1799 mg/m^2) by day 165 (Nov. 13), was a concentration comparable to most maxima in opaque columns during summer. The maximum Fe content occurred the same time as that of P - 3.4 mg/ℓ (2.97 g/m^2).

There was rather close agreement in the P release rates in eight of nine experiments in 1974. The values ranged from 2.6 to 4.4 mg/m^2 day for total P with a mean of 3.4. The one high value of 12.6 mg/m^2 day was not included because it was thought excessive mixing during sampling caused that high value. There was no difference in release rates between transparent or opaque columns once DO had been reduced to 2 mg/ℓ or below.

Table 16 shows a comparison of all values of release of P by various investigators in Lake Sammamish. The only values that seem excessively out of line are those of Monahan (1974) in the spring which were later shown to be accounted for by excessive mixing induced during sampling.

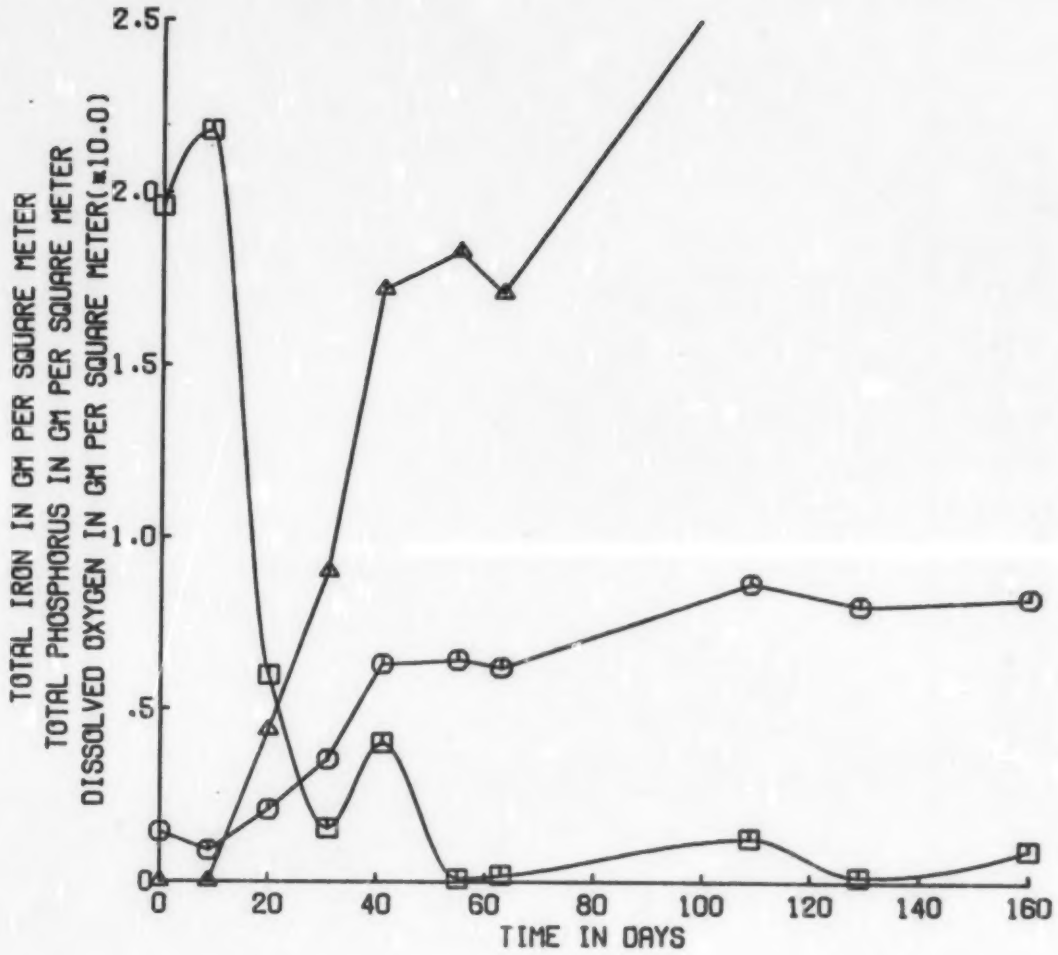
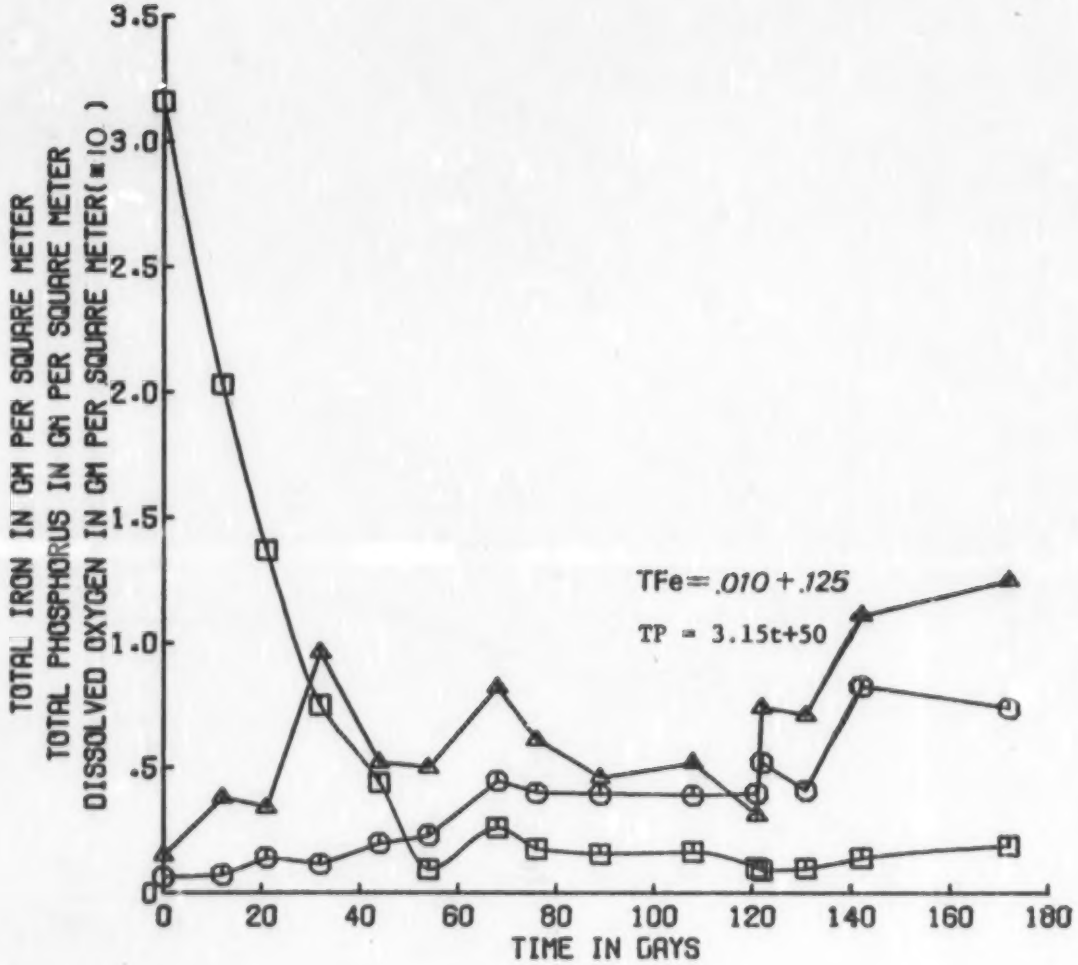


Figure 17. Changes in Total Iron and Total Phosphorus in an Opague Column from 8 August to 16 December 1974 (The column was partially de-oxygenated by nitrogen aeration). From McDonnel (1976).



□ DAYS VERSUS DO

○ DAYS VERSUS TP

△ DAYS VERSUS TFe

Figure 18. Changes in Total Iron and Phosphorus for an Opague Column from 27 July 1974 to 8 January 1975. (KH_2PO_4 - 0.236 mg/l was added on day 120). From McDonnell (1976).

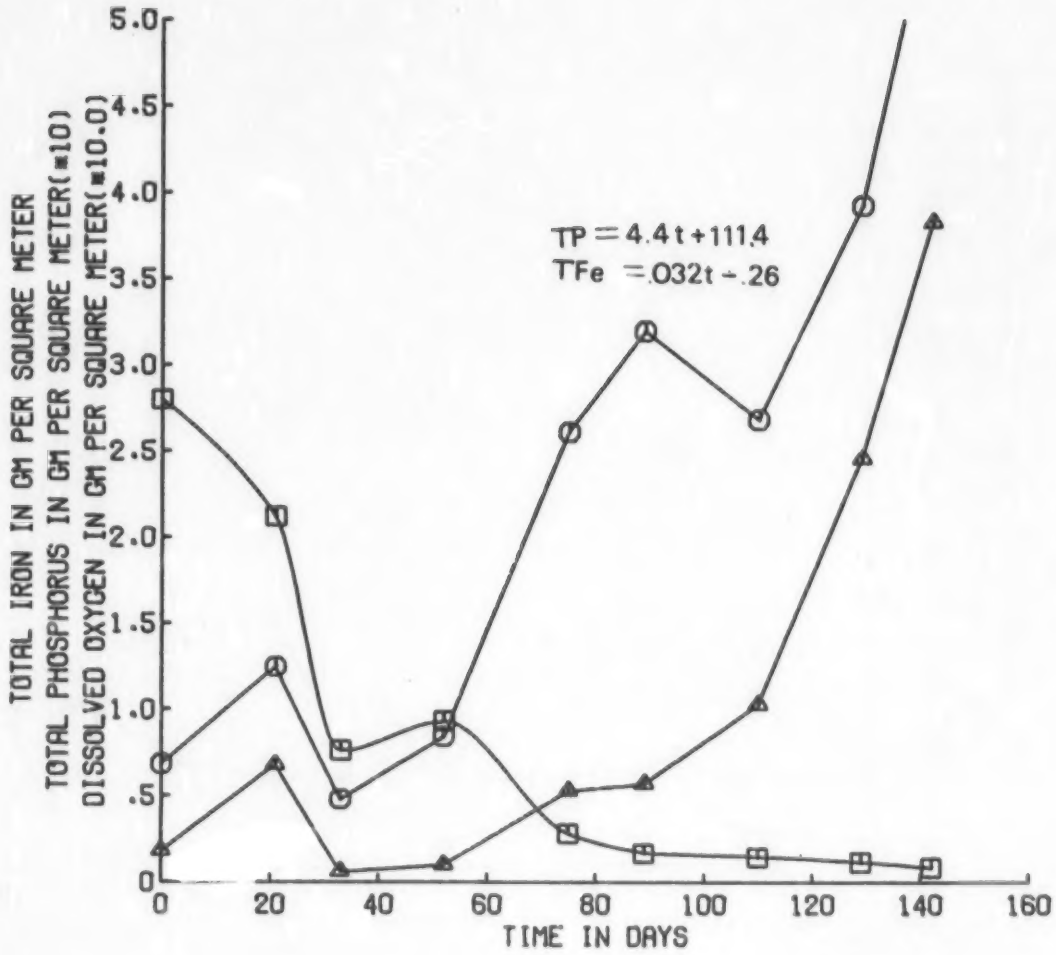
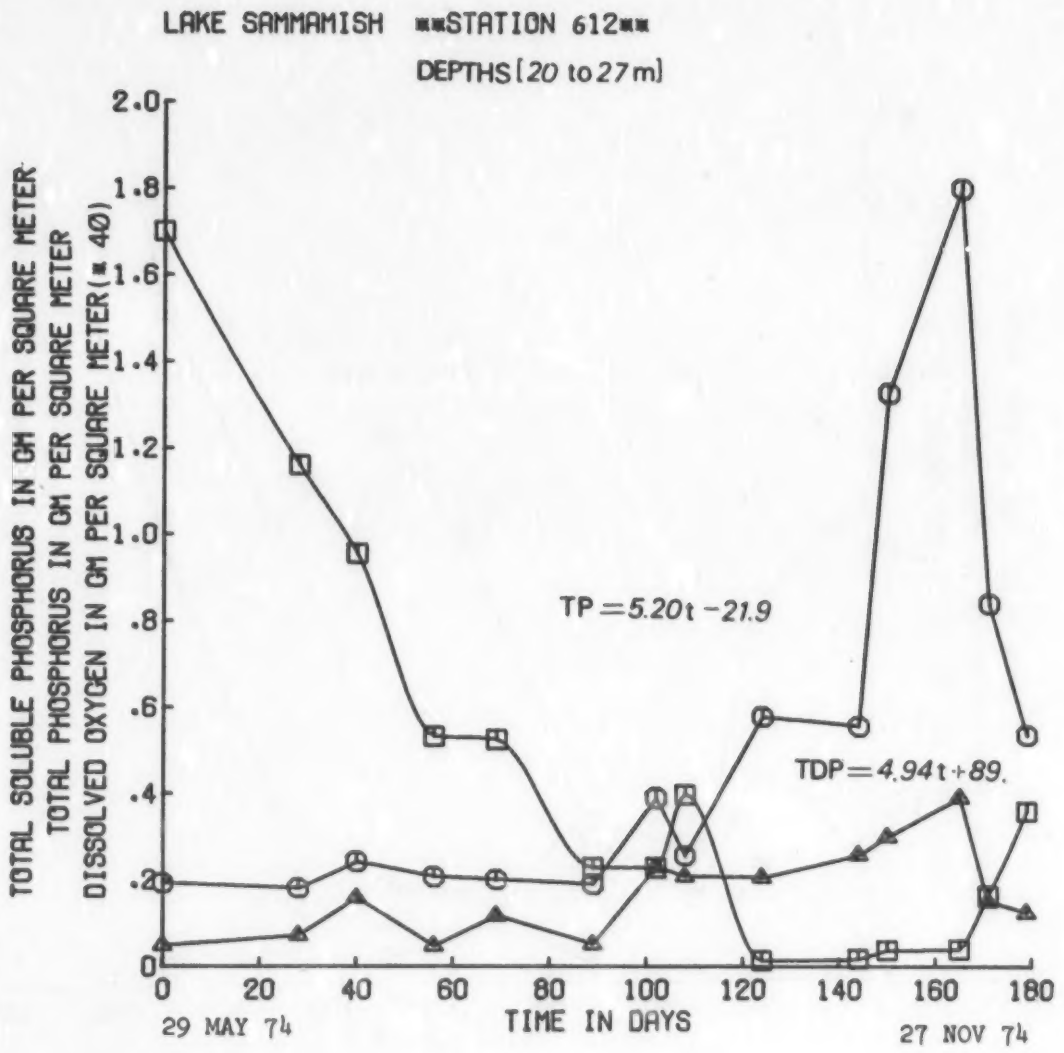


Figure 19. Changes in Total Iron and Phosphorus in a Transparent Column from 17 July 1974 to 8 January 1975. From McDonnell (1976).



⊡	DAYS	VERSUS DO	14 VALUES
⊙	DAYS	VERSUS TP	14 VALUES
▲	DAYS	VERSUS TDP	14 VALUES

Figure 20. Changes in Total and Soluble Phosphorus in the Lake Hypolimnion, Station 612, from 29 May to 27 November 1974. (Day 140 illustrates the commencement of the annual fall overturn). From McDonnell (1976).

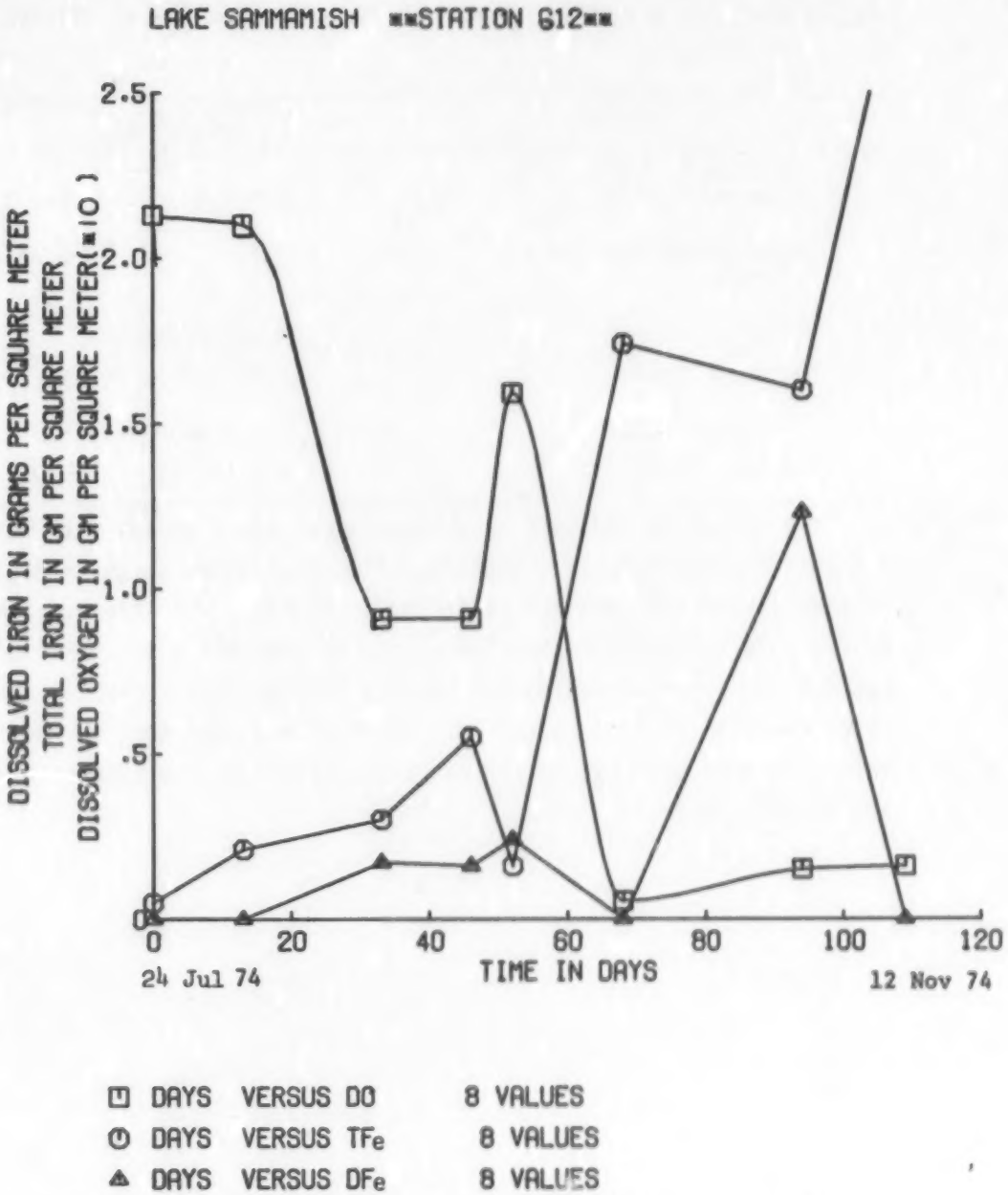


Figure 21. Release of Total and Soluble Iron in the Lake Hypolimnion, Station 612, from 29 May to 27 November 1975. From McDonnell (1976).

Table 16. A COMPARISON OF P RELEASE RATES FROM *in situ* COLUMN EXPERIMENTS AND OBSERVED HYPOLIMNETIC CHANGES IN CONCENTRATION DURING 1972-1975.

	Total P mg/m^2 day	Soluble P
in situ columns McDonnell (1975)	2.6-4.4	1.9-4.3
hypolimnion Station 612 1974	5.3	4.9
in situ columns Monahan (1974)	4.0-6.0 (fall) 17.0-26.0 (spring)	
in situ columns Horton (1972)	2.8 (summer) 6.2 (fall)	2.6 1.0

The effect of adding P to some columns (see Fig. 18) with respect to the maximum concentrations attained without addition suggests a situation of equilibrium and sediment buffering in the lake. P attained a maximum of 270 µg/ℓ following P addition to one column, while two other columns and the deep lake showed maxima between 260-280 µg/ℓ. Overall maxima ranged between 100-300 µg/ℓ. Near 300 µg/ℓ may represent an equilibrium value for the anaerobic bottom water assuming adequate contact with bottom sediments occurs.

MODELLING

Temperature profile output from the model is presented in Figures 22a through 22d for days when corresponding observed data were available for the years 1970-1973. These figures show that the simulated data give a close fit to the observed during this period.

Figure 23 gives the simulated mixing depth for March-July of 1971 and 1972. When this is compared to the epilimnion chl *a* values given in Figure 24 it may be noted that the earlier peak for the spring diatom pulse related closely to the consistently greater mixing depths the model had simulated. This output has caused us to reevaluate the hypothesis that the years of early blooms also have more light available to mixing diatoms because of a shallower mixing depth. The model suggests that mixing, which effects continued nutrient entrainment, may be more important than light availability. Another question raised about the 1971 spring diatom bloom concerned the source of PO_4-P being utilized. The PO_4-P pool, supply thought to be available was not sufficient to create a bloom of the magnitude of that which actually occurred. The model was able to simulate a bloom of similar magnitude only when fed PO_4-P from an external source. Diffusion from the lake bottom was eliminated as the major source because of the enormous diffusion coefficient which the model indicated would be required to effect the transfer. Littoral release and surface runoff are other possible sources to the PO_4-P pool to consider. The source from surface runoff is quite possibly a solution in the 1971 case since there was prolonged rainfall and subsequent runoff shortly before the spring bloom began.

Another example of information generated by the model is the fact that a sensitivity analysis of the model parameters showed that if the light intensity required for maximum phytoplankton production was held constant, as first hypothesized (Tang, 1975), the sensitivity of the model to that parameter was much greater than could be expected in the lake. Accordingly, it was felt that the function in that form was not an accurate representation and was therefore changed to its present

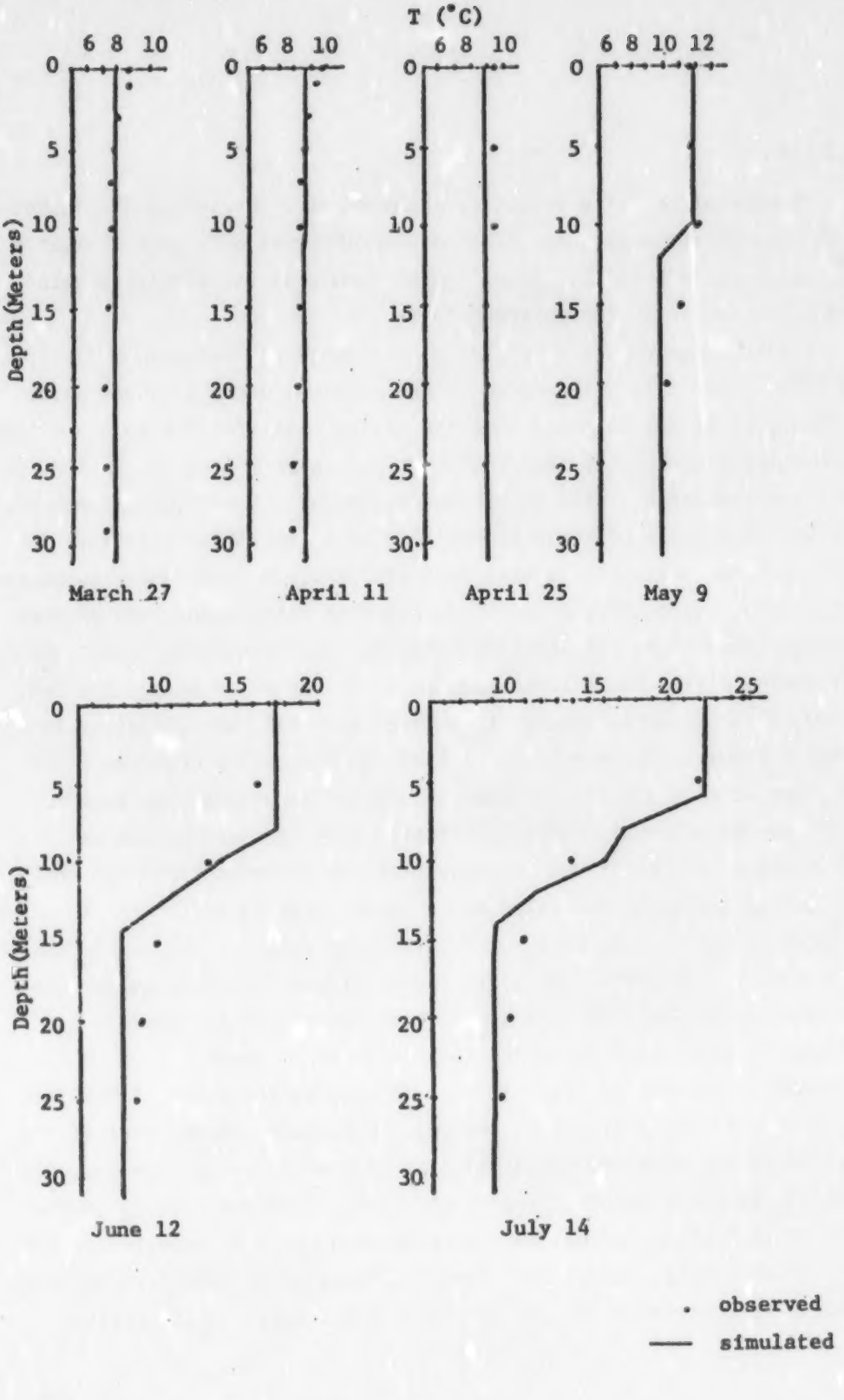


Figure 22. Simulated and Observed Temperature Profiles for Lake Sammamish (1970).

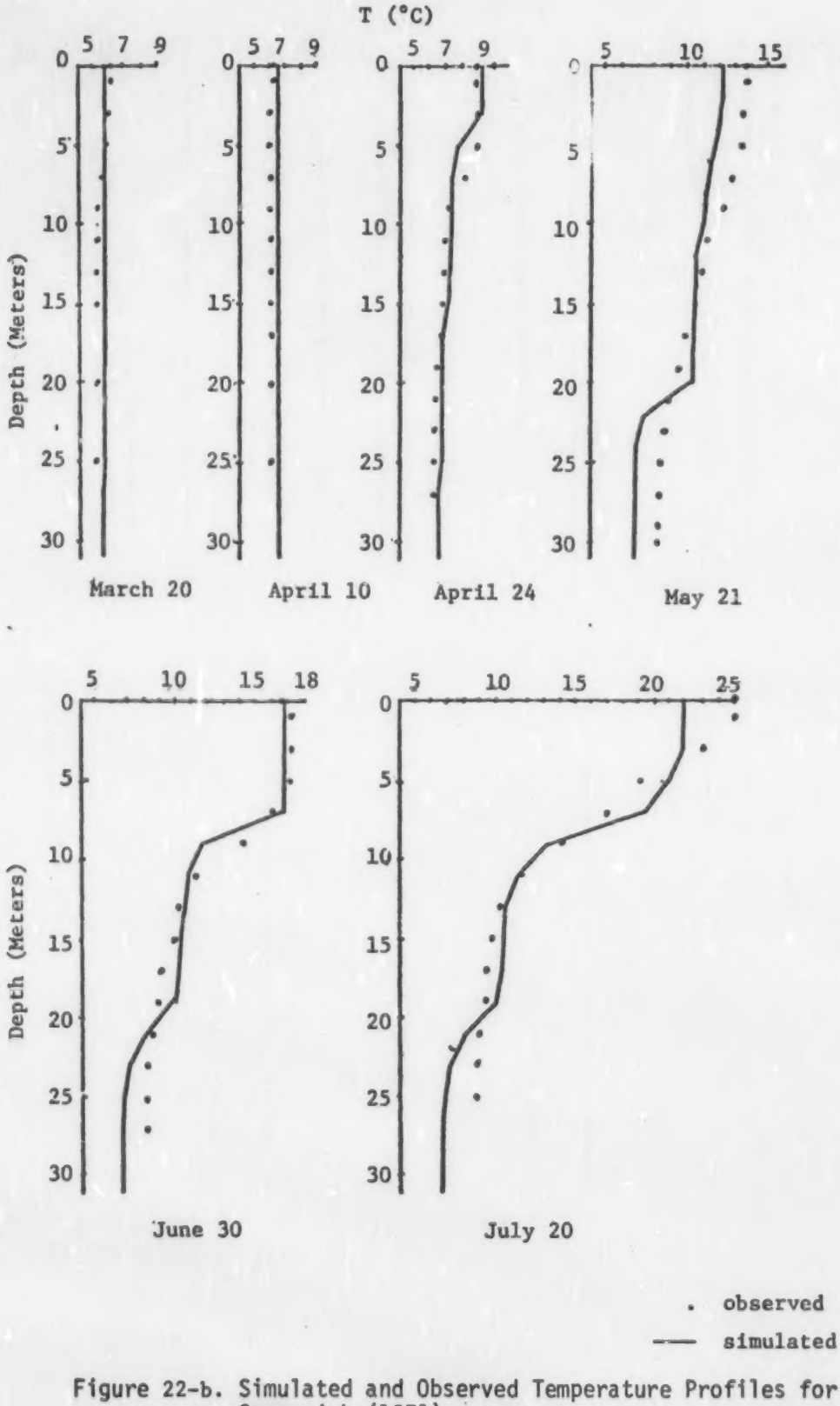


Figure 22-b. Simulated and Observed Temperature Profiles for Lake Sammamish (1971).

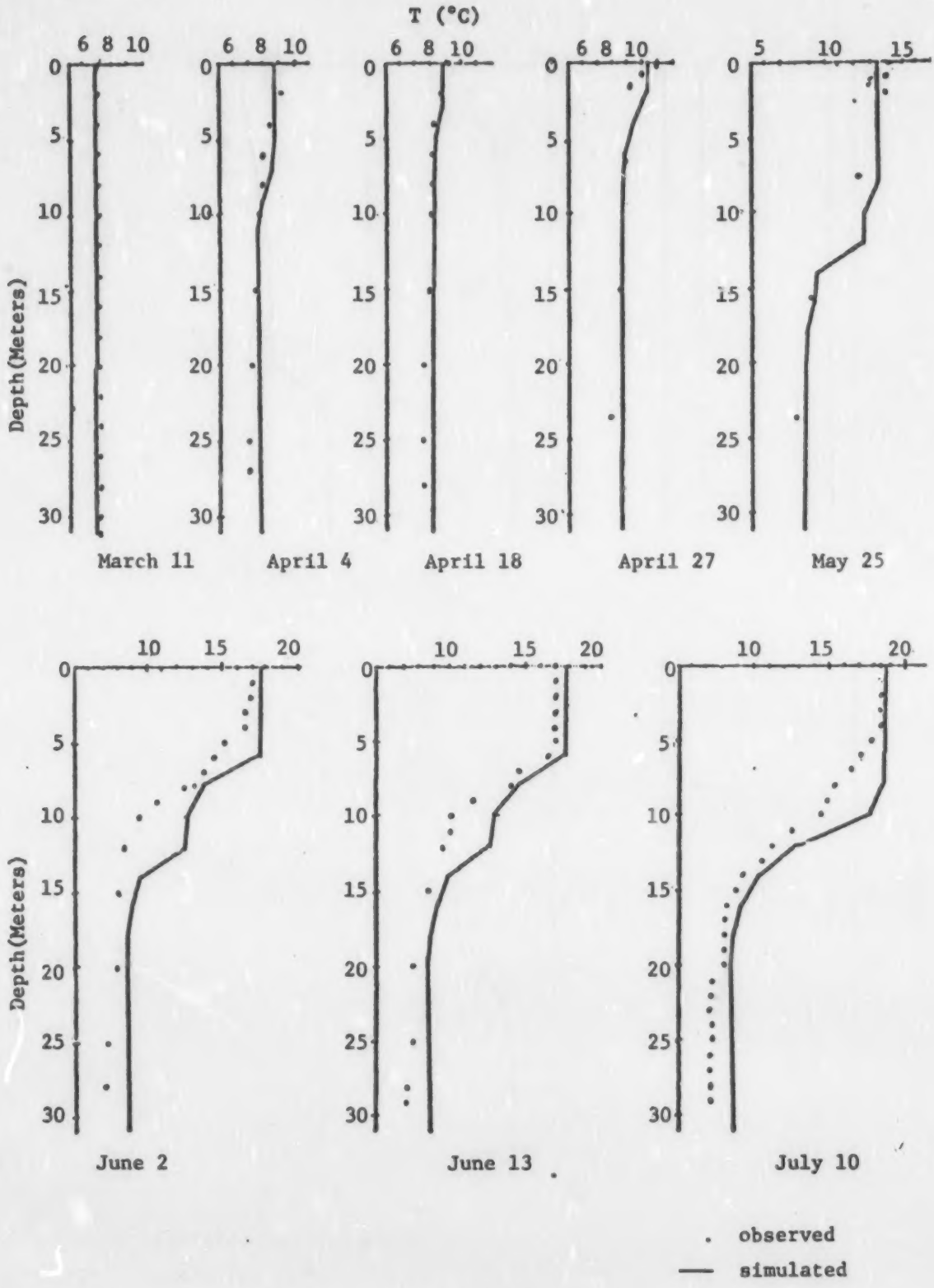


Figure 22-c. Simulated and Observed Temperature Profiles for Lake Sammamish. (1972)

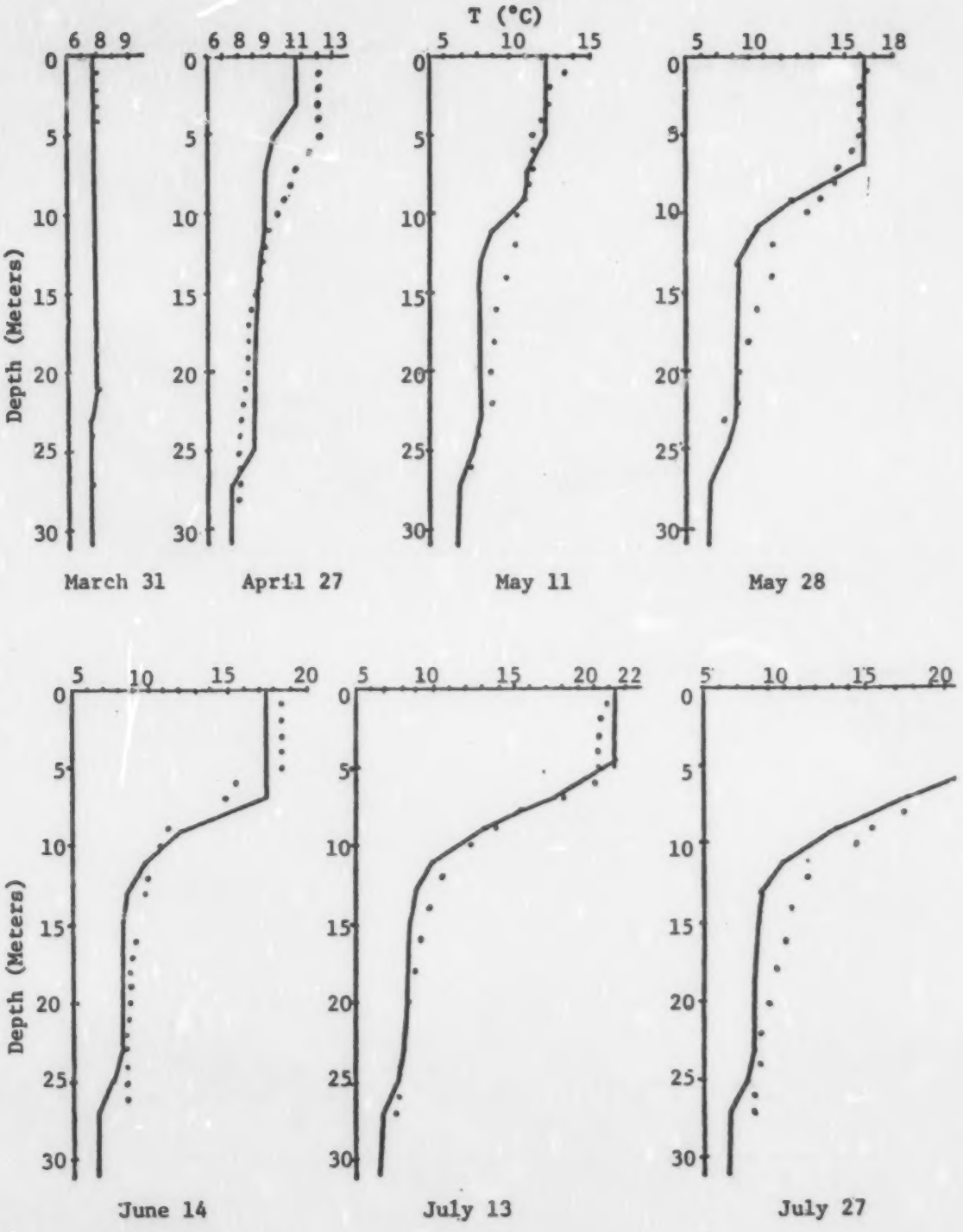


Figure 22-d. Simulated and Observed Temperature Profiles for Lake Sammamish (1973).

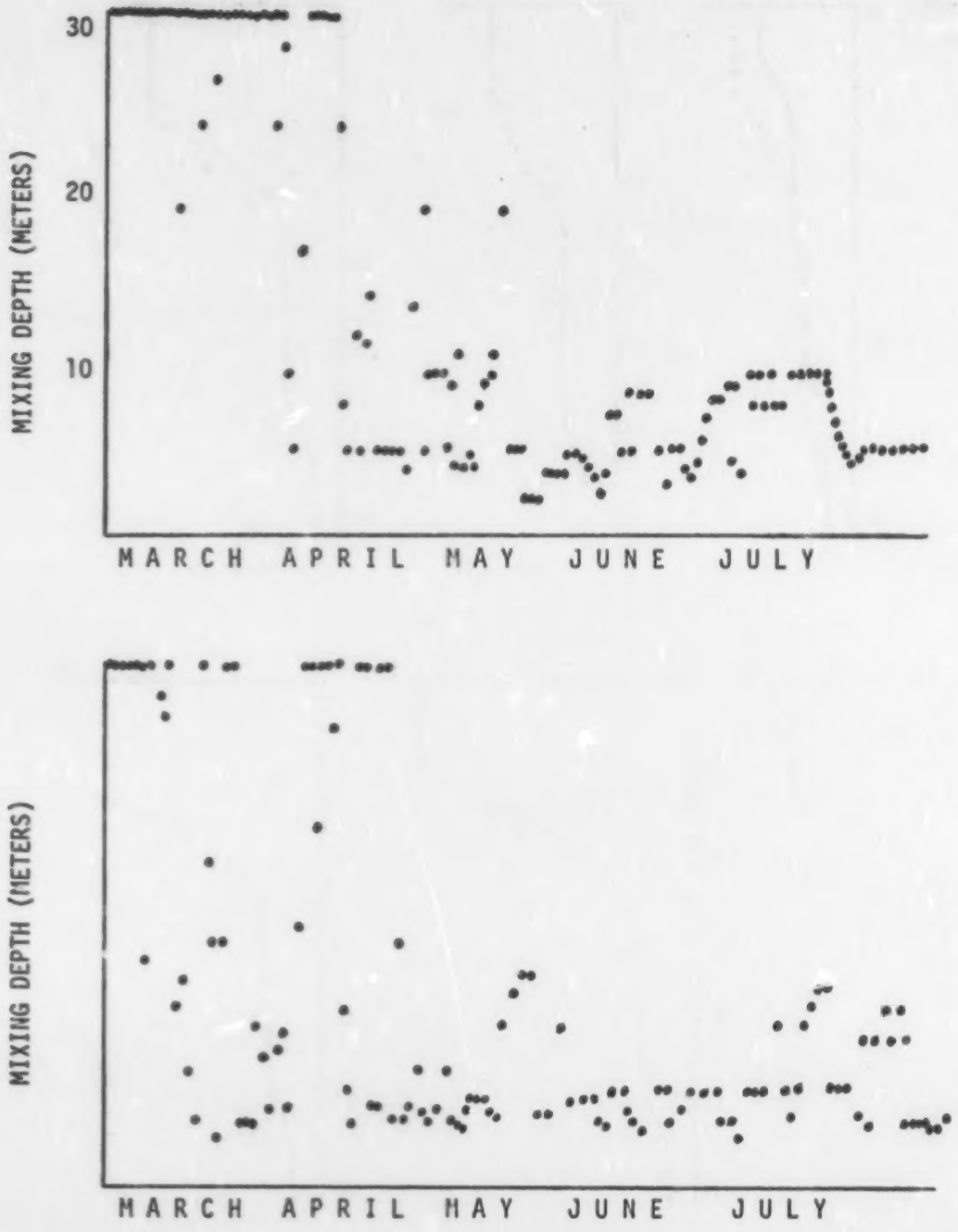


Figure 23. Calculated daily mixing depth in Lake Sammamish during March-July, 1971-72 (from Tang, 1975).

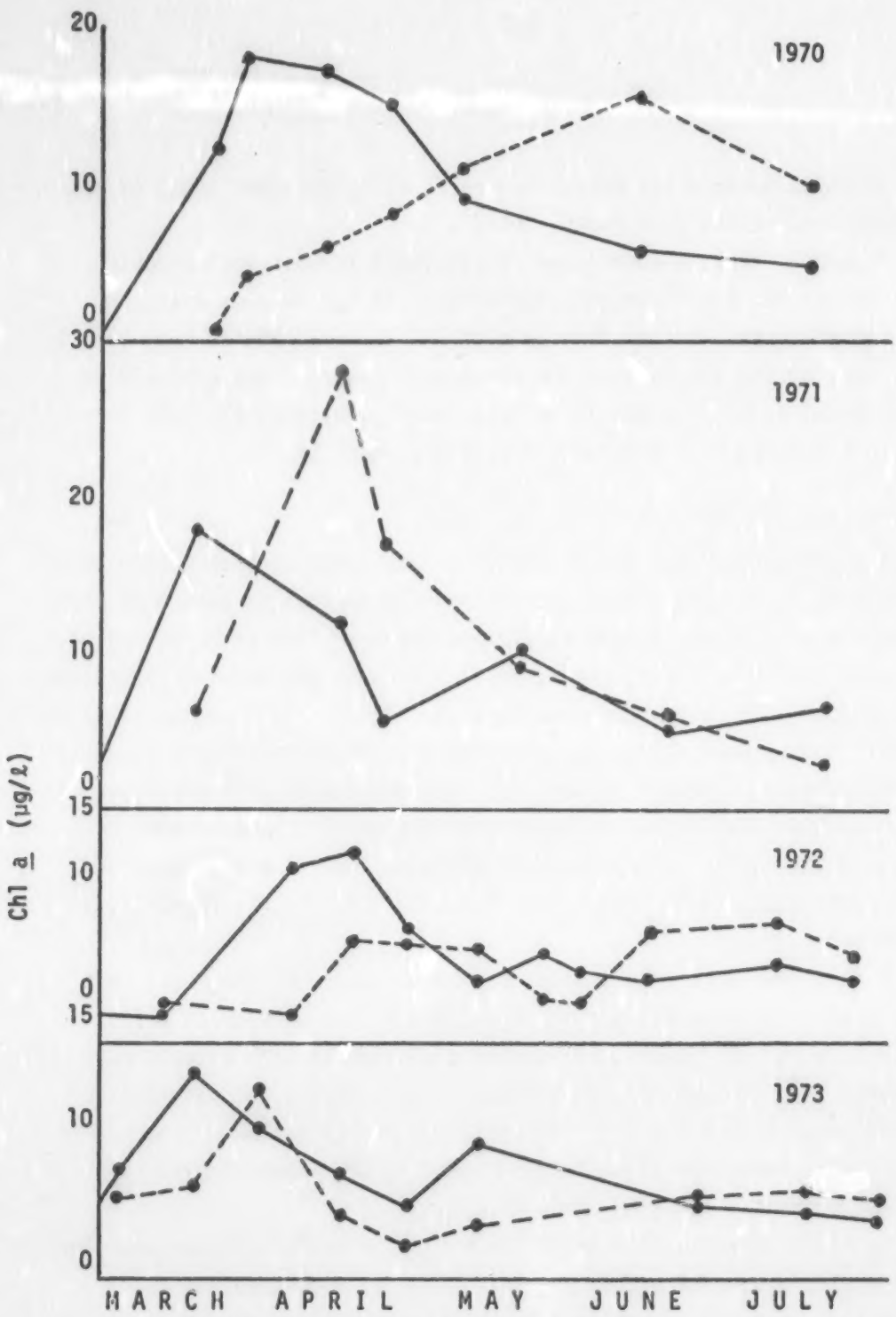


Figure 24. Simulated and observed chl a content in the epilimnion of Lake Sammamish during the springs of 1970-73.

form in which maximum production occurs at fifty per cent (50%) of the incident light level (see PHOTO, Table 8).

Figure 25 shows a comparison of simulated to observed concentrations in the PO_4-P pool in the epilimnion. It can be seen that, although the model does not simulate the PO_4-P pool exactly from year to year, some of the simulations are reasonably accurate and well within experimental error. It should be noted that no attempt was made to calibrate the model to provide the best fit possible.

MODEL OF THE NITROGEN CYCLE

The above-described multiparametric lake model considers both light and phosphorus as potentially growth-limiting factors to Lake Sammamish phytoplankton. There is some suspicion, however, that nitrogen may well have some effect on growth rates, particularly in the event of increased loading of P. In order to determine when nitrogen might be the limiting nutrient and to what extent its role would be in controlling productivity, and biomass, a nitrogen model has been developed which approximates the concentrations of the two major inorganic forms, ammonia and nitrate(Noah, 1976). These forms of N are not only the most abundant forms of N, but also the most significant in terms of plankton algal growth rate.

The following major processes can be evaluated by the model on a daily basis: (1) assimilation of inorganic N by phytoplankton, (2) denitrification of nitrate, (3) fixation of ammonia from molecular nitrogen, (4) lake inflow and outflow concentrations, (5) mineralization of ammonia from the detritus pool, (6) nitrification of ammonia to nitrate, (7) reduction of nitrate to ammonia, (8) input of N in rain, and (9) sediment release of ammonia.

In order to evaluate some of these rate processes, it was necessary to determine dissolved oxygen concentrations as a controlling factor. Rather than entering a complex array of concentrations over time and depth as exogenous input, an oxygen model was developed. The reactions

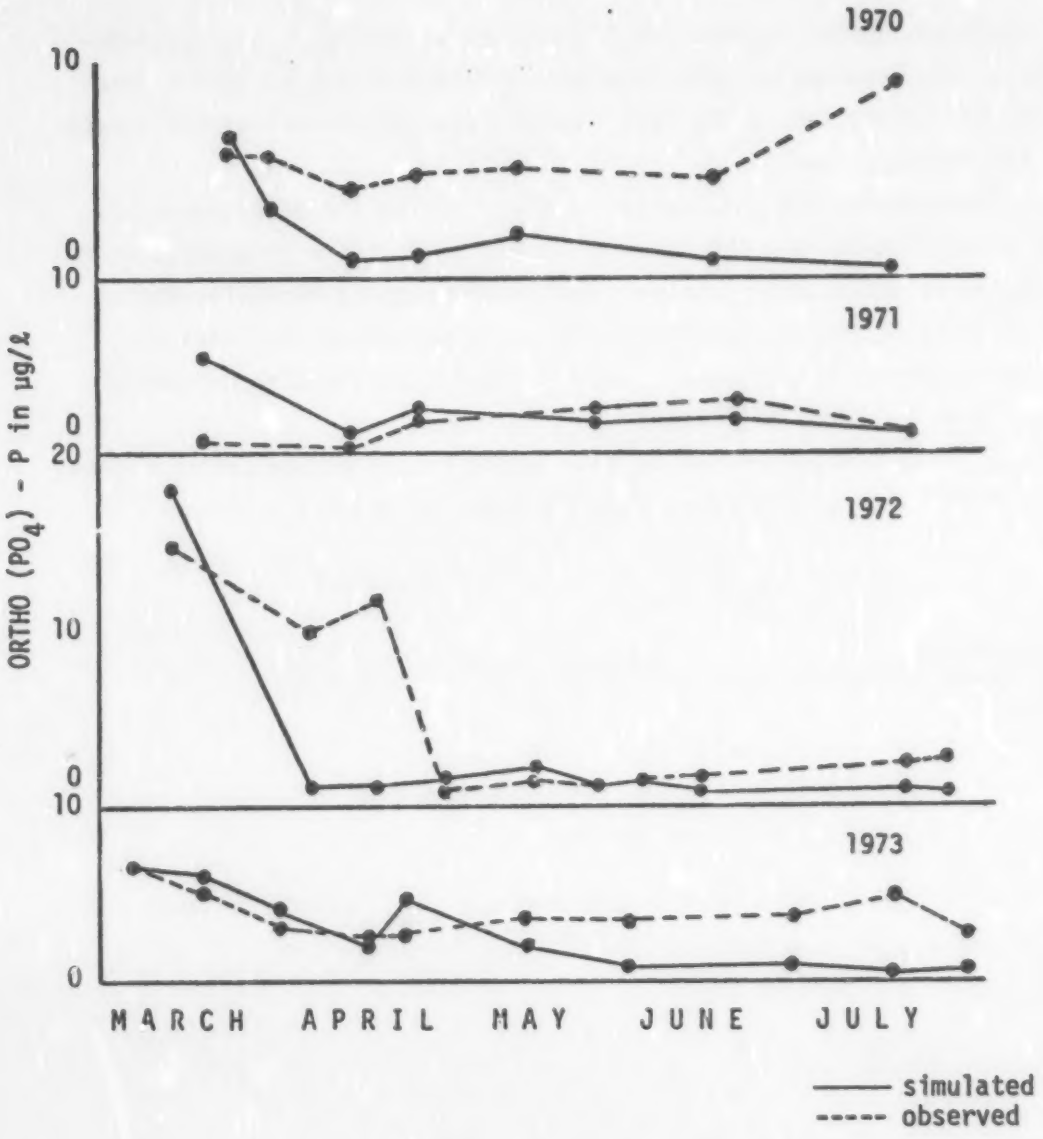


Figure 25. Simulated and observed ortho-P concentrations in the epilimnion of Lake Sammamish during the springs of 1970-73.

on which the system depends are (1) production of oxygen via photosynthesis, (2) consumption according to a biochemical oxygen demand function, (3) reaeration on the lake surface, and (4) benthal oxygen demand at the sediment layer.

Coordination and information transfer during the development of the nitrogen model and the sensitivity analysis of the phosphorus model revealed an interesting problem. Both models used a Michaelis-Menton function to limit phytoplankton growth. It was discovered that the Michaelis-Menton model has an apparent anomaly at limiting concentrations when used in a difference equation model.

As described by Mar (1976)[1] the anomaly occurs at low concentrations of nutrient and large biomass when the equation used is

$$\mu = \mu_s \left(\frac{S}{K_q + S} \right) \qquad \text{(equation 1)}$$

where μ = growth rate = $\frac{\text{productivity/unit time}}{\text{biomass}} = \frac{\Delta B}{B}$

S = concentration of nutrient

μ_s = growth rate of infinite concentration

K_q = 1/2 saturation constant

If $S >> K_q$ then μ approaches μ_s and when $S << K_q$ μ approaches zero. It is at these latter concentrations that the problem occurs. If $S << K_q$ then equation 1 reduces to

$$\mu = \frac{\Delta B}{B} = \mu_s \left(\frac{S}{K_q}\right)$$

or $$\Delta B = \mu_s \left(\frac{S}{K_q}\right) B$$

This equation says that the change in biomass (ΔB) for a given concentration of substrate is a function of how much biomass in present. This means that no matter how little substrate is available the amount of

[1]Manuscript in review, ASCE.

growth it can support can be increased by increasing the amount of biomass assimilating it. This is clearly incorrect and in a difference equation model causes an overestimation of growth if the nutrient concentration approaches zero.

A new form of the equation has been proposed by Mar (1976) which is of the form

$$\mu = \mu_s \left(\frac{S}{\alpha B + S} \right)$$

As S approaches zero this equation reduces to

$$\mu = \frac{\Delta B}{B} + \mu_s \left(\frac{S}{B}\right)$$

or

$$\Delta B = \mu_s \left(\frac{S}{\alpha B} \right) B = \frac{\mu_s S}{\alpha}$$

This means that growth is dependent only upon the concentration of substrate at these values while at values where $S >> \alpha B$. The equation reduces to the same form as the Michaelis-Menton function.

This new equation has been incorporated in the nitrogen and phosphorus models with the result output of chl a for 1972 as shown in Fig. 26. The maximum growth rate was also reduced by 75% to delay the ch/s increase.

Comparison of this information to that provided by the earlier phosphorus model (Fig. 24) shows a much closer fit of the simulated data to the observed data. Because of its more accurate simulation the new equation is considered a better approach to practically deal with modelling of nitrogen and phosphorus dynamics in lakes.

The model has not shown nitrogen to limit phytoplankton growth during the March-to-July period and under the existing P loading regime. Nitrogen may indeed prove to be temporarily limiting growth rate, or at least have an interactive role, in late summer or early fall and/or under higher P loading rates. Such extended application of the N model was beyond the scope of this project, however. The formulation, parameter evaluation and validation of this model was described by Noah (1976).

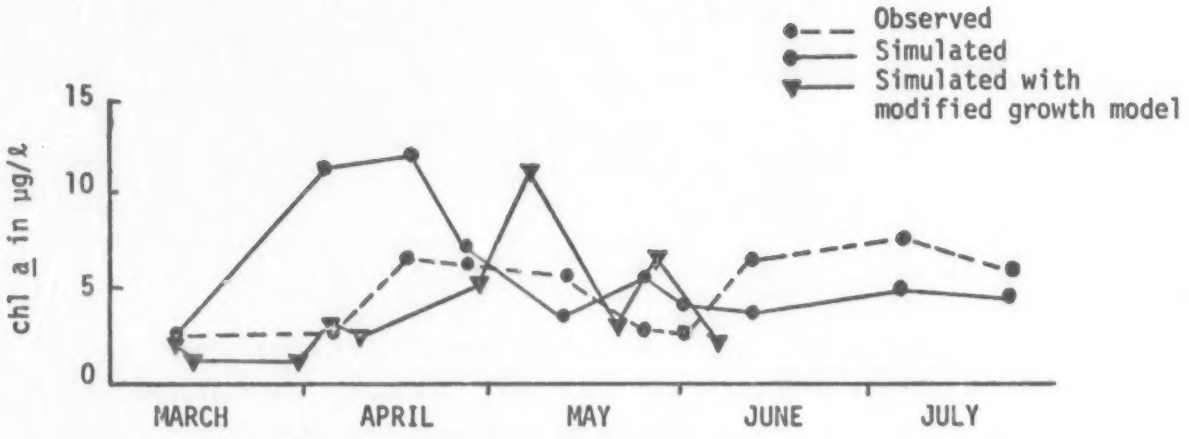


Fig. 26. Simulated and observed chl a content in the epilimnion of Lake Sammamish during the spring of 1972. Note relatively better simulation of the spring increase in chl a with the modified growth model.

SECTION 9
DISCUSSION

HYPOTHESIS FOR STABILITY

Lake Sammamish has shown no detectable changes in most of the pertinent lake quality variables in the six years since one third of the phosphorus income was diverted. Water clarity has remained unchanged since diversion - the mean summer Secchi disk depth is 3.3 m. Mean winter total phosphorus concentration has remained at about 30 µg/l ± 2 µg/l before and after diversion. Chlorophyll a, averaged over the growing season and throughout the photic zone, has remained at about 7 µg/l ± 2 µg/l. The only variable that may have changed is the fraction of blue green algae, which shows an average decrease of 46 percent compared to pre-diversion (1964-65) values.

The apparent stability in the trophic state of Lake Sammamish, in spite of the watershed settlement by European man beginning over 100 years ago and the diversion of about one-third of its phosphorus income in 1968, can be largely attributed to the sediment-water interchange mechanism for phosphorus. As much as three-fourths of the phosphorus increase into the water column during the stratified period in May-November could potentially be released as soluble P from the anaerobic sediments as shown in plastic column experiments. The moderately high and constant phosphorus content in the sediment profile along with the indication that the lake's hypolimnion has been anoxic in late summer for a long time suggests that the P release rate has always been large in Lake Sammamish. Water column balances of P for the turnover period show a greater maximum before diversion (5-9 g/m^2) than after diversion ($\bar{<}2$ g/m^2). Since there has been no change in the winter water column content of P (31 µg/l) since diversion, the excess at turnover in pre-diversion years 1964 and 1965 was most probably lost to sedimentation (Rock, 1974).

The release and sedimentation mechanism is thought to be largely controlled by iron. Phosphate is released in proportion to iron as oxygen content reaches low levels and is eventually exhausted in late summer. Just prior to lake turnover the water column content of dissolved P is at the annual maximum, most of which is in the hypolimnion. Instead of these high concentrations being maintained throughout the winter, much of that P is resedimented when turnover restores the aerobic conditions and P decreases in proportion to Fe. Presumably much of this P is complexed with iron. The relatively constant winter concentration of 30 µg/l following the fall turnover appears to be residual, the excess having been largely removed by the complexation with iron. Presumably the additional phosphorus income before diversion had not increased the pre-growth period residual P concentration because of the ample quantity of iron available for P complexation and sedimentation - up to 0.9 mg/l Fe at maximum anaerobiosis. The large prediversion quantities of P that appeared at fall turnover then could have come from accumulations of flocculent sedimented P, that is, the quantity of annual P income removed by Fe over and above the residual P level. The anaerobic nature of the hypolimnion in this lake then seems to behave as a greater effective sink for P than if the hypolimnion were aerobic. That is, the large quantities of Fe released under anaerobic conditions would not have been so released in the aerobic state (Fe in the Lake Sammamish hypolimnion is 10 times that in Lake Washington) and these would not be available for the precipitation of biologically mineralized P.

While complexation of hypolimnetic P with iron and sedimentation subsequent to turnover (Nov. - Dec.) is probably the principal mechanism controlling the residual level of P, another contributing mechanism operates during Jan - Feb. To account for the continual decrease in water column P content during those later winter months, the relatively large amount of P input during that period (runoff is usually high) is believed to be rapidly sedimented along with the large quantity of P reactive colloidal material brought in through the high flows. Secchi

disk measurements are lower during that period than any other time during the year and a large fraction of the material caught in sediment traps is clay, which together reflect the large contribution from land runoff.

Additionally, early spring blooms of diatoms remove P from the epilimnion more effectively than non-diatoms. Years in which early diatom blooms occur also have lower spring and summer mean P concentrations in the photic zone. This mechanism is not contributing to maintenance of the wintertime 30 µg P/l residual, but may contribute to resistance to increased P income of lakes with low mixing potential. Interestingly, the years with early spring diatom blooms show one-half the March - August productivity compared to non-early-bloom years.

Thus, it appears that in the range of P income to Lake Sammamish, 1.0 g $P/m^2 \cdot yr$ before to 0.67 g $P/m^2 \cdot yr$ after diversion, and assuming no change in the sediment release of P, the P content could have remained relatively unchanged with only a change in the sedimentation rate. In further support of this relative stability in that range of P income, a modified version of a mass balance model (Vollenweider, 1969) can be used to predict only a slight change in water column content of P (Krull, 1973; Welch, et al., 1973). Phosphorus input, outflow, and exchange with the sediments were included in the model. The lake was assumed to be two layered during the stratified period and uniformly mixed during the remainder of the year, and the water column phosphorus content was considered to be influenced only by the above mentioned gains and losses; complex cycling within the water column was ignored. The model considered sedimentation and release rate separately. The release rate was assumed constant, while the sedimentation rate was not held constant. This was the key departure from the Vollenweider model (Vollenweider, 1969) which earlier had shown that Lake Sammamish P should exhibit a prompt and significant recovery (Emery, et al., 1973). The modified model indicated only a slight change (7 percent) in phosphorus concentration to a one-third reduction in phosphorus loading.

Although the above model allowed the sedimentation rate to vary, the sedimentation rate varied proportional to the phosphorus concentration in the water column. Therefore, as an another approach to realistically reflect the lake dynamics the sedimentation rate Rock (1974) allowed to vary with the phosphorus loading to the lake. The second modified model was;

$$\frac{dP}{dt} = \frac{M}{A} - K_0 P + R - S \qquad (1)$$

where,

S = sedimentation rate = $K_1 \frac{M}{A}$

P = total phosphorus in the water column

R = release rate

M = input loading rate

A = surface area of lake

K_0 = output rate coefficient

K_1 = sedimentation rate coefficient

The release rate was assumed constant. Data showed that the winter column concentration of total phosphorus was unchanged from pre-diversion to post-diversion. Under these conditions the model could be calibrated. At pre-diversion steady state;

$$\frac{dP}{dt} = 0, \quad \frac{M}{A} - K_0 P = D$$

where,

D = retention rate of phosphorus in lake,

thus, equation (1) becomes;

$$D = S - R \qquad (2)$$

and for pre-diversion lake conditions equation (2) becomes;

$$\frac{20{,}000 - 3{,}300 \text{ kg}}{19.8 \text{ km}^2} = \frac{20{,}000 \text{ kg}}{19.8 \text{ km}^2} \cdot K_1 - R \qquad (3)$$

at post-diversion steady state, equation (2) becomes;

$$\frac{13{,}000 - 3{,}300 \text{ kg}}{19.8 \text{ km}^2} = \frac{13{,}000 \text{ kg}}{19.8 \text{ km}^2} \cdot K_1 - R \qquad (4)$$

equations (3) and (4) were solved by simultaneous equations, hence $K_1 = 0.96$.

Solution of equation (1) yields the change in steady state phosphorus concentration (ΔP) for a given step change in loading (ΔM)

$$P = \frac{\Delta M}{A} (1 - K_1) \qquad (5)$$

The solution of equation (5) for the 1968 sewage diversion (ΔM = 7000 kg P) with $K_1 = 0.96$ gives a P of only 0.014 g/m^2 in the water column. This represents a concentration change in the water column of only 0.5 µg P/l (~2%). Thus, the model indicates that if a decrease in loading was followed by a decrease in sedimentation rate, the water column phosphorus concentration should not have been significantly reduced.

IMPLICATION OF NON-RECOVERY

Although non-recovery would suggest that the sewage diversion project was a failure, such a conclusion is completely unwarranted. The primary objective of the project was to stop the cultural eutrophication of Lake Sammamish. Although the trophic state in terms of lake concentrations was not reversed, the present trophic state does not represent deteriorating nor intolerable conditions. In fact, the mean summer chl a level in Lake Sammamish is now about equal to that found in Lake Washington following diversion (Edmondson, 1972). Further, summer water clarity, as measured by the Secchi disk has been greater than 3 meters on the average during all the years investigated, before and after diversion. Lake Washington on the other hand showed average summer Secchi disk depths of only 1 m before diversion (Edmondson, 1972) and now has similar clarity as Sammamish.

Although Lake Sammamish did not noticeably improve following diversion of wastewater, the action was no doubt a preventative measure. The lake may have in fact advanced in trophic status without the 1968 diversion. The lack of response over the past 100 years to the range of P income of 1.0 to 0.67 g $P/m^2.yr$ tempts one to speculate that future increase may also not affect trophic state. That is probably not tenable because although most of the above increase in P probably came in the last 10 years. Issaquah, the main source of sewage, has nearly tripled in population since 1960 and the wastewater from the Issaquah area has doubled since diversion in 1968. This suggests that the increase in P income to Lake Sammamish over the past century has been in the "lag phase" and only very recently could the increase be described as entering the "growth phase" of an exponential function. Without diversion, the P income would not be 0.67 g $P/m^2.yr$, or 1.0 g $P/m^2.yr$ (the pre-diversion rate), but rather 1.36 g $P/m^2.yr$, which would reflect the recent growth in the Issaquah area since 1968. Such an increase would most probably not all go into the sediments and the present winter mean content of 31 µg/l would surely have increased. This seems especially reasonable since the iron supply for complexation would not have increased proportionately.

Although Lake Sammamish has remained stable over a range in a P income of 0.67 to 1.0 g $P/m^2.yr$ that probably has little implication for future increases in income. The important question now relates to the expected increase from suburban and urban runoff that can be expected as the forested areas around the lake, particularly the east side, are reduced to residential areas. At first approximation the yield of P from such developed areas in the Sammamish watershed do not appear great, but in light of what has been demonstrated by careful analysis of the P transport in Issaquah Creek (high runoff rates yield high P concentrations) the question of yield increase from residential development must receive more careful study if a totally reliable prediction is to be obtained.

The implication of this lake's response to P income manipulation

to the management of other lakes is not entirely clear. Certainly one is left with the overriding impression that lakes with anaerobic hypolimnia are more unpredictable than those that remain aerobic. If the degree of P manipulation in Sammamish is compared with that of Washington on the Vollenweider graph (Figure 27) it becomes of course clear that the degree of manipulation was greater in Washington. Yet from that graph one would have expected a proportional change in lake trophic state, other factors being equal, since the loading in Sammamish was decreased to a point near the "excessive" line similar to that in Washington. Obviously such a proportional response did not occur, thus one must agree with Vollenweider and Dillon (1974), for anaerobic lakes at least, that P removal in excess of 50% of the input is necessary to observe a significant recovery. In support of that Larson, et al. (1975) have observed an improvement in the trophic state in Shagawa Lake, which has an anaerobic hypolimnion, as a result of an interception of 70% of the P income (Fig. 27). However, Shagawa was very eutrophic at a similar loading as Sammamish (Malueg, et al., 1973). Clearly more of the incoming P in Lakes Washington and Shagawa ended up in the water column and available to algae than was the case in Lake Sammamish. The waste water P entered Lakes Washington and Shagawa directly while in Sammamish the waste water entered through Issaquah Creek. This could be at least part of the reason why much of the loaded P to Sammamish does not end up in the water column but rather in the sediment. Issaquah Creek carries a rather high load of iron so possibly much more of the P added to Issaquah Creek was destined for the sediments of Lake Sammamish than was the case for the waste water P added to the other two example lakes.

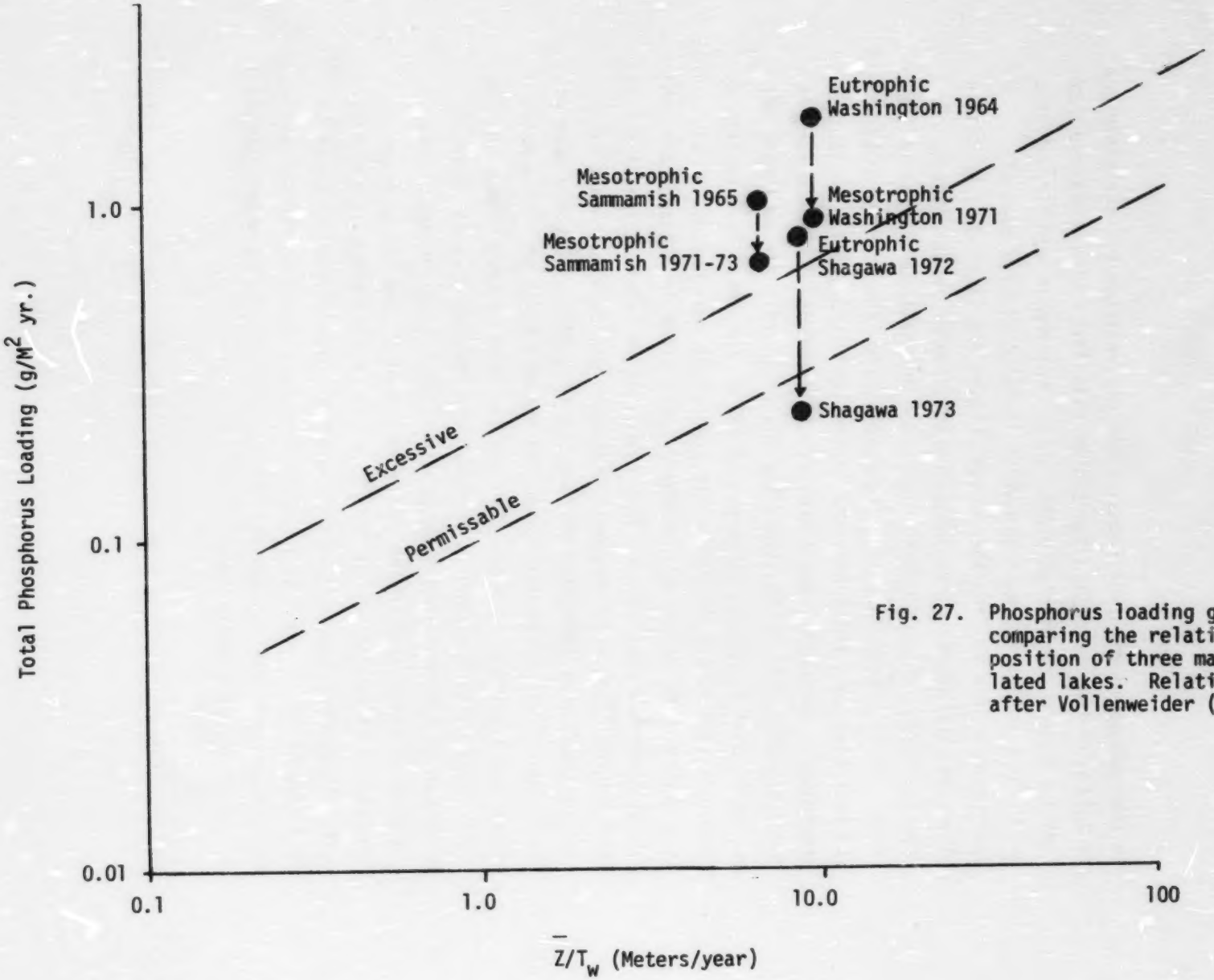


Fig. 27. Phosphorus loading graph comparing the relative position of three manipulated lakes. Relationship after Vollenweider (1974).

CONTRIBUTION OF THE MODEL TO UNDERSTANDING

As the results have shown, the model has provided some interesting information as well as raising questions which have provided impetus and direction for new or more thorough research in certain critical areas. As with any modeling effort its major contributions are as a test vehicle for new hypotheses and as a source of direction for research which will expand understanding.

This model has helped to increase understanding in such areas as the importance of mixing of Lake Sammamish waters to algal growth, the effect of light intensity on algal growth, the relative importance of diffusion in the water column versus external inputs of nutrients and effects of wind and insolation upon themixed depth of the lake.

Interesting questions have been raised as to littoral and bottom release as contributers to the PO_4-P pool as well as the effects of surface runoff and atmospheric fallout. The model is available for use in testing any new theories which are generated in these or other areas of interest using Lake Samamish as the study site.

SECTION 10

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